FINAL
Remedial Investigation Report
IR Site 2, West Beach Landfill and Wetlands
Alameda Point, California

Volume I

Prepared for

Base Realignment and Closure
Program Management Office West
1455 Frazee Rd., Suite 900
San Diego, CA 92108

CONTRACT NUMBER: N68711-01-D-6009
TASK ORDER NUMBER: 0007

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June 2006

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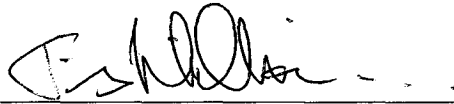# ALAMEDA POINT, CALIFORNIA

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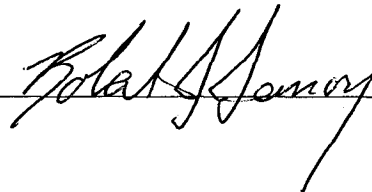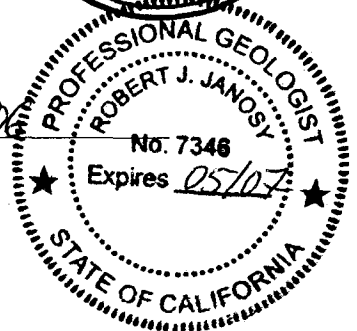Travis Williamson, P.E. _____   Date 6/22
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EXECUTIVE SUMMARY

This report documents the Remedial Investigation (RI) implemented at the West Beach Landfill and Wetlands located at the former Naval Air Station (NAS) Alameda in Alameda, California as part of the RI/Feasibility Study (FS) process. The West Beach Landfill and Wetlands are collectively identified as Installation Restoration (IR) Site 2, which coincides with Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) Operable Unit (OU) 4A. The purpose of this RI Report is to document characterization and investigation methods employed at the site, provide and summarize the data generated and data interpretation and evaluation completed, and develop conclusions regarding the nature and extent of contamination and the human health and ecological risk posed by contamination.

Site Characterization and Previous Investigations

IR Site 2 encompasses roughly 110 acres in the far southwestern portion of Alameda Point, with the landfill portion of the site occupying approximately 77 acres and the wetlands occupying approximately 33 acres. The site was originally constructed in a shallow open water and intertidal environment through dredging and filling. Beginning in approximately 1956, IR Site 2 was used as the principal disposal location for wastes generated at the former NAS Alameda. Landfilling at IR Site 2 was reportedly terminated in early 1978. Over the course of its operation as a landfill, IR Site 2 reportedly received up to 1.6 million tons of general base waste as well as a number of specific and potentially hazardous industrial or process waste streams, including, but not limited to, asbsestos, solvents, waste oils, pesticides, and sandblasting grit. Disposal occurred primarily in the landfill portion of the site, although historical information suggests that some minor amount of disposal could have occurred in limited portions of the wetlands. Historical information suggests that dredge material from other sites at former NAS Alameda was placed in the wetland area at IR Site 2. Following cessation of landfill activities at IR Site 2, landfill closure activities were initiated. However, these closure activities were not comprehensively completed.

Numerous investigative activities historically have been completed at IR Site 2 to evaluate the type and extent of contamination and to study overall ecological health at the site. These activities have included:

- Phases 1 and 2A Solid Waste Assessment and Testing (SWAT) activities conducted in 1990;

- Phases 5 and 6 SWAT activities conducted in 1991;

- An ecological assessment conducted in 1993;

- Wetland evaluation technique (WET) analysis conducted in 1993;

- Field activities conducted in support of an ecological assessment in 1994 and 1995;

- Radiological surveys conducted from 1995 to 1999;

- Supplemental ecological investigations conducted in 1996 and 1997;

- Regular groundwater monitoring conducted beginning in 1991;

- Biological sampling conducted in support of an ecological risk assessment in 1998;

- An ordnance and explosive waste (OEW) survey and removal action conducted in 2002 and 2003; and

- Geotechnical and seismic evaluations conducted in 2002 and 2003.

A comprehensive RI sampling plan was implemented at the site during two seasonal sampling events in 2004 and 2005 to address data gaps from previous investigations and to generate a robust site-specific database related to the overall nature and extent of contamination at IR Site 2. To develop a fully representative dataset of environmental conditions at IR Site 2 during the RI, soil, groundwater, and tissue were sampled in the landfill portion of the site and soil, groundwater, sediment, surface water, and tissue were sampled in the wetland portion of the site. One round of sampling was completed during the dry season of 2004 (October 2004) and a second round of sampling was completed during the wet season of 2005 (March 2005). In general, data were generated from areas within the footprint of the landfill and within the inundated and non-inundated portions of the wetlands, as these areas largely had been left uncharacterized by historical investigations. Geophysical and radiological surveying were implemented to assess site conditions and to guide the RI, and exploratory trenching was conducted to specifically evaluate subsurface conditions at the site. To aid in distinguishing impacts at IR Site 2 from ambient environmental conditions, background data were generated at Alameda Point and ambient environmental conditions were assessed at portions of a local reference station (China Camp State Park [CCSP]) with characteristics similar to the site but not affected by site activities or potential site-related contamination. Toxicity and bioaccumulation studies were conducted on site and reference media. Over the course of the RI, hundreds of individual samples of various environmental media were collected, providing thorough coverage of the site and its distinct habitat types. A comprehensive suite of compound classes was analyzed for in samples of various environmental media at IR Site 2 and the selected reference locations, including metals, volatile organic compounds (VOCs), polychlorinated biphenyls (PCBs), pesticides, semivolatile organic compounds (SVOCs)/polycyclic aromatic hydrocarbons (PAHs), polychlorinated dibenzodioxins (PCDDs)/polychlorinated dibenzofurans (PCDFs), explosives, radionuclides, and petroleum hydrocarbons.

Based on the geophysical surveying, cover soil in the landfill is underlain by material with a widespread and diffuse pattern of electromagnetic response indicative of large volumes of historically disposed waste. The geophysical surveying was not able to resolve specific waste types or discrete disposal locations given the extensive electromagnetic response throughout the landfill. The wetland portion of the site appears to be largely free of historically disposed waste. Limited exploratory trenching conducted in the landfill portion of the site confirmed the presence of a wide variety of waste material in the subsurface (e.g., paper, plastic, and wood), excluding characteristically hazardous or imminently dangerous material such as OEW, drums, cylinders, and radiological waste.

Nature and Extent of Contamination

In combination with appropriate and usable historical data, the RI site characterization data were evaluated to assess the overall occurrence and distribution of contamination at IR Site 2. Data were separated by different areas of the site (landfill and wetland), media types (soil, groundwater, surface water, and sediment), and discrete intervals where applicable (e.g., surface versus subsurface soil and first water-bearing zone versus second water-bearing zone groundwater), and evaluated using statistical and graphical techniques. The analytical results indicate that a number of individual compounds were detected in the various media sampled at IR Site 2. The RI generally demonstrates that numerous compounds are present in a widespread manner at the site and that, with few exceptions, compounds do not exhibit patterns or trends indicative of contamination hotspots or clear and discrete source areas. Many compounds detected in media at IR Site 2 exhibit concentrations very similar to or even lower than concentrations of these compounds in the background dataset and/or ambient environment, suggesting that the site and historical operations at the site are not a likely source of these compounds and that the potential risk posed by such compounds is entirely consistent with the risk posed by background/ambient conditions. Alternately, a number of other individual compounds detected in media at IR Site 2 exhibit concentrations significantly higher than concentrations of these compounds in the background dataset

and/or ambient environment, suggesting that the site and historical operations conducted at the site are potentially a source of these contaminants.

Overall, subsurface soil is more highly impacted compared to surface soil across the site, and the landfill is more highly impacted than the wetland portion of the site. Data indicate that groundwater at the site is not substantially impacted by contamination. Geologic conditions at the site, which include a generally continuous hydrogeologic confining unit below the shallowest groundwater zone and a deeper confining unit that isolates the deeper aquifer system beneath Alameda Point, further mitigate the potential for widespread groundwater impacts. In addition, although there is some evidence of variability in the occurrence and distribution of contamination in the two surface water bodies in the wetland portion of the site between seasons and also between the water bodies, these features do not appear to be substantially impacted by the presence of any type of contamination.

Certain individual compounds detected in media at IR Site 2 appear to be present in what could be considered localized hotspots. For instance, lead was detected in soil at two locations at the site at concentrations notably higher than all other locations. Also, certain VOCs in groundwater appear to demonstrate at least some type of plume behavior, although maximum concentrations of these compounds are not significantly elevated. However, with the exception of a fairly limited number of compounds in the various environmental media assessed at the site, the data do not suggest clear source areas or significant contaminant hotspots, but rather indicate a widespread and diffuse occurrence of various types of compounds.

Human Health and Ecological Risk Assessments

A human health risk assessment (HHRA) and an ecological risk assessment (ERA) were implemented to evaluate the potential for adverse human health effects or adverse effects to the natural environment and ecological receptors from exposure to contaminants at the site. In accordance with guidance from the United States Environmental Protection Agency (U.S. EPA), the U.S. Navy (Navy), and the California Department of Toxic Substances Control (DTSC), the HHRA and ERA for IR Site 2 were performed using a tiered process. The first tier for each was a screening-level assessment that included the development of a preliminary conceptual site model (CSM) focusing on pathways between site contamination sources and human or ecological receptors, identification of contaminants of potential concern, and a screening-level assessment of contaminant exposure and human health effects or a screening-level dose response assessment. The second tier for each was a baseline assessment that formulated specific exposure scenarios, defined specific exposure assumptions, and more thoroughly screened contaminants of potential concern. In the baseline assessment, exposures and effects were modeled for a number of specific endpoints, and human health or ecological risks were quantified for each of these endpoints.

For the HHRA, the specific exposure scenarios included exposure to site media via direct contact, incidental ingestion, inhalation, ingestion, and/or external radiation for a set of appropriate hypothetical receptors that included site workers (a park ranger/tour guide, a park ranger/restoration supervisor, and a construction/excavation worker) and a site visitor. For the ERA, the specific exposure scenarios included exposure to site media via direct contact, incidental ingestion, and/or root contact for a set of representative receptors from ecological categories including plants, terrestrial and aquatic invertebrates, mammals, birds, fish, and burrowing animals.

The HHRA concluded that one metal (arsenic), several SVOCs/PAHs (benzo(a)pyrene, benzo(k)fluoranthene, dibenz(a,h)anthracene, and naphthalene), two pesticides (dieldrin and delta-HCH), total PCBs, PCDDs/PCDFs, and two radionuclides (radium-226 and radium-228) are potential risk drivers for at least one human receptor class evaluated in the landfill and/or wetland. Potential risks from these compounds

were determined to be related to their presence in surface soils, subsurface soils, groundwater, and/or surface water. Radium-228 also was determined to be a potential risk driver to at least one human receptor class, but the presence of this compound is more likely to be the result of natural processes (i.e., the decay of naturally-occurring thorium-232) rather than historic site activities. Specific pathways found to be of potential concern during the HHRA of the landfill and wetland include direct dermal contact with surface soil, shallow groundwater, or surface water; incidental ingestion of surface soil; inhalation of vapors from surface or subsurface soil; and exposure to external radiation from surface or subsurface soil.

The ERA concluded that several metals (antimony, arsenic, cadmium, chromium, copper, lead, manganese, mercury, molybdenum, nickel, selenium, silver, vanadium, and zinc), several SVOCs/PAHs (acenaphthene, chrysene, fluoranthene, fluorene, naphthalene, phenanthrene, total low-molecular-weight PAHs [LPAHs], and total high-molecular-weight PAHs [HPAHs]), total PCBs, several pesticides (*alpha*-chlordane, dieldrin, *gamma*-chlordane, *trans*-nonachlor, and total DDx [i.e., the sum of DDT, its congeners, and its derivatives]), and PCDDs/PCDFs are potential risk drivers to at least one ecological receptor in the landfill, wetland, and/or wetland pond portions of the site. Of these compounds, chromium, lead, mercury, nickel, total HPAHs, and total DDx were identified as the most significant risk contributors (i.e., exhibit the highest hazard quotients [HQs]) for various portions of the site. Contaminants were evaluated based on surface soil sampling data for mammals, birds, and terrestrial invertebrates. For plants, contaminants were evaluated based on surface and subsurface soil sampling data. For the wetland ponds, contaminants were evaluated based on surface water and sediment sampling data for all receptors. Specific pathways found to be of potential concern at the site based on the ecological risk assessment include direct dermal contact with surface soil or sediment; incidental ingestion of surface soil, sediment, or surface water; and root contact with surface or subsurface soil.

Several of the compounds determined to be potential risk drivers at IR Site 2 were characterized in background media at Alameda Point and in ambient media at the nearby CCSP reference station. Moreover, several of the compounds determined to be potential risk drivers at IR Site 2 were actually found at highly consistent or even higher concentrations in the background/ambient media datasets. Specifically:

- Mean detected concentrations of arsenic, chromium, copper, lead, manganese, nickel, vanadium, benzo(a)pyrene and benzo(k)fluoranthene were higher in CCSP upland soil compared to surface soil from the landfill portion of IR Site 2.

- Maximum detected concentrations of chromium, manganese, nickel, vanadium, benzo[a]pyrene, and benzo[k]fluoranthene were higher in CCSP upland soil compared to surface soil from the IR Site 2 landfill.

- Mean and maximum concentrations of nickel and vanadium were higher in CCSP upland soil compared to subsurface soil from the landfill portion of IR Site 2.

- Mean detected concentrations of arsenic, chromium, copper, manganese, molybdenum, nickel, selenium, vanadium, zinc, dieldrin, and total DDx were higher in CCSP wetland soil compared to surface soil from the wetland portion of IR Site 2.

- Maximum detected concentrations of manganese, molybdenum, vanadium, and total DDx were higher in CCSP wetland soil compared to surface soil from the IR Site 2 wetland.

- Mean detected concentrations of chromium and vanadium were higher in CCSP wetland soil compared to subsurface soil from the wetland portion of IR Site 2.

- The maximum detected concentration of vanadium was higher in CCSP wetland soil compared to subsurface soil from the IR Site 2 wetland.

- Mean and maximum detected concentrations of arsenic, vanadium, *gamma*-chlordane, dieldrin, *trans*-nonachlor, and total DDx generally were higher in CCSP sediment compared to surface sediment in both of the IR Site 2 surface water bodies.

- Mean and maximum detected concentrations of chromium, copper, lead, mercury, nickel, and zinc were higher in CCSP sediment compared to surface sediment in one of the IR Site 2 surface water bodies.

Accordingly, the potential environmental risk associated with the presence of these compounds in the ambient environment would be expected to be consistent with or even higher than the risk associated with the presence of these compounds in media at IR Site 2. In addition, the findings of toxicity tests implemented during the RI using media collected directly from IR Site 2 suggest that there is no potential for toxicity to benthic invertebrates, water column invertebrates, or fish resulting from constituents present in IR Site 2 media. Both unfiltered and filtered aliquots were generated for most aqueous samples collected during the RI. However, in developing the HHRA, only data from the unfiltered samples were used. Given that aqueous samples were collected from temporary well points and shallow surface water bodies during the RI, a significant amount of entrained turbidity was generally present in the samples. Given the tendency of contaminants like PCBs, PCDDs/PCDFs, and metals to remain sorbed to solid matrix material, it is possible that conclusions regarding the risk posed to humans by contaminants in shallow groundwater or surface water are overly conservative. Nevertheless, no compounds concluded to be potential risk drivers through the HHRA or ERA have been discounted based on their ambient concentrations and associated ambient risk, their lack of toxicity as concluded by toxicity assays, or the bias associated with the use of generally turbid water samples.

Contaminant Fate and Transport

Mechanisms of contaminant migration that are potentially important at the site include direct movement of contamination in solid matrix material, dissolution and movement in overland runoff, dissolution and movement in groundwater, volatilization, and movement in the surface water system. These mechanisms represent not only the most likely modes of contaminant transport in the future, but also the most likely explanations for the current occurrence and distribution of contamination in environmental media at the site following cessation of historical site operations. In addition, some physical and biological processes likely have been and continue to be important in determining the occurrence and distribution of contamination at IR Site 2, including sorption of generally hydrophobic compounds such as PCBs, volatilization of readily volatile compounds, biodegradation of certain compounds, and radioactive decay of certain radionuclides.

Summary and Conclusions

Given the magnitude of the RI implemented at IR Site 2 and the sheer volume of information available from historical investigations and this RI, the Navy has focused this RI Report on thoroughly evaluating and describing the nature and extent of site contamination and developing appropriate human health and ecological risk assessments to determine specific contaminants that are potential risk drivers at the site. The next stages for remediation at IR Site 2 will include the development of remediation goals, the assessment of suitable remediation strategies, the selection of an appropriate remediation plan, the development of a remedial design, and, ultimately, the implementation of a site remedy. As such, upon finalization of this RI Report, the Navy will develop a thorough FS that will include the definition of remediation goals, the establishment of a required remediation footprint, and the evaluation of remedial

strategies. The FS will thoroughly consider all appropriate risk management requirements within the context of the calculated site risks, the types of contaminants responsible for risk, the distribution of potential risk drivers in the environment, and proposed site redevelopment plans.

Risk inputs to the FS also will be considered in light of key sources of uncertainty identified in the HHRA and ERA in this RI Report. These uncertainties are directly relevant to the utility of the conclusions of the risk assessments in the context of risk management decision-making. Among the various sources of uncertainty identified in this RI Report, uncertainty regarding the appropriate definition of background and ambient conditions is one of the most important. Due to the common use of conservative assumptions to mitigate uncertainty when developing risk assessments, findings of potentially unacceptable risk at or below background or ambient concentrations are common. However, it is very important that risk management contemplate the role of background and ambient risk in the ultimate site remediation framework. Any remedial options considered in subsequent phases of assessment must consider ambient or background sources in the context of the potential for recontamination following remediation efforts and the contribution of background/ambient risk. Regardless, constituents have not been removed from further consideration in this RI Report based on comparisons to ambient or background concentrations.

Based on planned future use at IR Site 2 and ultimate resolution of ambient/background conditions, it is anticipated that a presumptive remedy approach (i.e., landfill capping) would be effective at mitigating any unacceptable risks that are present at site. The Navy looks forward to evaluating this alternative among other potentially acceptable alternatives during the Feasibility Study phase of the project.

CONTENTS

APPENDICES (provided on CD-ROM)

ABBREVIATIONS AND ACRONYMS

a	intercept of regression model
A_c	area of the site
A_R	surface area of contaminated road segment
A_s	area of the site
A_{site}	area of the site
ABS	absorption factor
ABS_d	dermal absorption factor
ABS_{GI}	gastrointestinal absorption factor
ACF	area correction factor
AE	assessment endpoint
AF	soil-to-skin adherence factor
AF_a	soil-to-skin adherence factor, adult
AF_c	soil-to-skin adherence factor, child
ALM	adult lead model
ARRA	Alameda Reuse and Redevelopment Authority
ASTM	American Society for Testing and Materials
AT	averaging time
AT_c	averaging time for carcinogens
AT_n	averaging time for non-carcinogens
AT/AP	anti-tank/anti-personnel
ATSDR	Agency for Toxic Substances Disease Registry
AWQC	ambient water quality criteria
b	slope of regression
B	ratio of the permeability of the stratum corneum relative to the permeability across the viable epidermis
BAAQMD	Bay Area Air Quality Management District
BAF	bioaccumulation factor
BBL	Blasland, Bouck & Lee, Inc.
BCG	biota concentration guideline
BERA	baseline ecological risk assessment
bgs	below ground surface
BRA	baseline risk assessment
BRAC	Base Realignment and Closure Act (of 1990)
BSU	Bay Sediment Unit
BTAG	(U.S. EPA Region 9) Biological Technical Assistance Group
BW	body weight
BW_a	adult body weight
BW_c	child body weight
C_{air}	adjusted exposure concentration in outdoor air
$C_{building}$	indoor air concentration
$C_{groundwater}$	concentration in groundwater
C_{prey}	concentration in prey tissue
$C_{prey\ 1,2,3}$	concentration in each food type
C_{sed}	concentration in sediment
C_{soil}	concentration in soil
$C_{soilgas}$	concentration in soil gas
C_w	concentration in water

°C	degrees Celsius
CARB	California Air Resources Board
CCME	Canadian Council of Ministries of the Environment
CCR	California Code of Regulations
CCS	California Coordinate System
CCSP	China Camp State Park
CDFG	California Department of Fish and Game
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act
CF	conversion factor
CFR	Code of Federal Regulations
COD	chemical oxygen demand
COPC	contaminant of potential concern
COPEC	contaminant of potential ecological concern
CPT	cone penetrometer testing
CR	contact rate (or ingestion rate)
CSF	cancer slope factor
CSF_d	cancer slope factor for dermal contact
CSM	conceptual site model
CTE	central tendency exposure
cy	cubic yard
D_i	diffisivity in air
D_w	diffisivity in water
DA	apparent diffusivity
DA_{event}	absorbed dose per event per area of skin exposed
DCA	dichloroethane
DCCO	double-crested cormorant
DCE	dichloroethene
DDD	dichlorodiphenyldichloroethane
DDE	dichlorodiphenyldichloroethene
DDT	dichlorodiphenyltrichloroethane
DDx	sum of the total DDD, DDE, and DDT concentrations
DEQ	Department of Environmental Quality
DF_i	dilution factor for indoor air inhalation
DGM	digital geophysical mapping
DO	dissolved oxygen
DOE	United States Department of Energy
DON	United States Department of the Navy
DQO	data quality objective
DRO	diesel-range organics
DTSC	(California) Department of Toxic Substances Control
E&E	Ecology and Environment, Inc.
ED	exposure duration
ED_a	adult exposure duration
ED_c	child exposure duration
EF	exposure frequency
EOD	explosive ordnance disposal
EPC	exposure point concentration
ERA	ecological risk assessment
ER-L	effects-range low

ER-M	effects-range median
ET	exposure time per day
ET_o	exposure time fraction, outdoors
ET_i	exposure time fraction, indoors
EV	event frequency
F_D	dispersion correction factor
f_{oc}	fraction of organic carbon in soil
FA	fraction absorbed water
FI	fraction of soil ingested that is contaminated
FIFRA	Federal Insecticide, Fungicide, and Rodenticide Act
FOD	frequency of detection
FS	Feasibility Study
FWBZ	first water-bearing zone
FWEC	Foster Wheeler Environmental Corporation
GI	gastrointestinal (tract)
GIS	geographic information system
GRO	gasoline-range organics
GSD_i	geometric standard deviation
GSF	gamma shielding factor
H'	Henry's Law Constant
HEAST	(U.S. EPA's) Health Effect Assessment Summary Tables
HHRA	human health risk assessment
HI	hazard index
HMX	octahydro-1,3,5,7-tetranitro-1,3,5,7-tetrazocine
HPAH	high-molecular-weight PAH
HpCDD	heptachlorodibenzodioxin
HPGe	portable high-purity Germanium system
HQ	hazard quotient
HRG	Habitat Restoration Group
IAEA	International Atomic Energy Agency
IAS	Initial Assessment Study
IEUBK	(U.S. EPA) Integrated Exposure Uptake Biokinetic
IFS_{adj}	age adjusted soil ingestion factor
InR_{adj}	age-adjusted inhalation factor
InR_a	adult inhalation rate
InR_c	child inhalation rate
IR	Installation Restoration
IR_a	adult ingestion rate
IR_c	child ingestion rate
IR_{food}	estimate of daily ingestion rate of food
IR_{soil}	soil ingestion rate
$IR_{soil/sed}$	estimate of receptor's daily incidental ingestion rate of soil or sediment
IR_x	ingestion rate
IRP	Installation Restoration Program
IRIS	Integrated Risk Information System (U.S. EPA)
ITSI	Innovative Technical Solutions, Inc.

K_d	soil-water partition coefficient
K_{oc}	organic carbon partitioning coefficient
K_{ow}	octanol-water partitioning coefficient
K_p	skin permeability coefficient of a compound in water
KSR	Kister, Savio, and Rei, Inc.

LCL	lower confidence limit
LOAEL	lowest observable adverse effects level
LOEC	lowest observed effect concentration
LPAH	low-molecular-weight PAH

MARSSIM	Multi-Agency Radiation Survey and Site Investigation Manual
MCL	maximum contaminant level
MDL	method detection limit
ME	measurement endpoint
MEK	methyl ethyl ketone (2-butanone)
MGV	molar gas volume
mph	miles per hour
MTBE	methyl-*tert*-butyl ether
MW	molecular weight
μCi	microCuries

n	total soil porosity
N/A	not applicable
NA	not available/not analyzed
NAD	North American Datum
NAVFAC SW	Naval Facilities Engineering Command, Southwest
NAS	Naval Air Station
NC	not calculated
NCRP	National Council on Radiation Protection and Measurement
ND	not detected
NEC	no effect concentration
NGVD	National Geodetic Vertical Datum
NOAA	National Oceanic and Atmospheric Administration
NOAEL	no observable adverse effects level
NOEC	no observed effect concentration
NTCHS	National Technical Committee for Hydric Soils
NTU	nephalometric turbidity units
NWS	Naval Weapons Station

OCDD	octachlorodibenzodioxin
OEHHA	Office of Environmental Health Hazard Assessment
OEW	ordnance and explosive waste
ORNL	Oakridge National Laboratory
OU	Operable Unit

P	wetland pond
PAH	polycyclic aromatic hydrocarbon
PbB	blood lead
PCB	polychlorinated biphenyl
PCE	tetrachloroethene

pCi	picocurie
PCDD	polychlorinated dibenzodioxin
PCDF	polychlorinated dibenzofuran
PEF	particulate emission factor
PEL	probably effect level
PID	photoionization detector
ppbv	parts per billion by volume
ppmv	parts per million by volume
ppt	parts per thousand
PRC	PRC Environmental Management, Inc.
PRG	preliminary remediation goal
PWC	(Navy) Public Works Center
QAO	Quality Assurance Officer
QC	quality control
Q/C	inverse of the mean concentration at the center of a square source
Q/C_{sa}	inverse of the mean concentration at the center of a 0.5-acre square source
Q/C_{sr}	inverse of 1-h average air concentration along a straight road segment bisecting a square site
Ra-226	$Radium^{226}$
Ra-228	$Radium^{228}$
RBSL	risk-based screening level
RDX	hexahydro-1,3,5-trinitro-1,3,5-triazine
RfC	reference concentration
RfD	reference dose
RfD_d	reference dose for dermal contact
RI	Remedial Investigation
RI/FS	Remedial Investigation/Feasibility Study
RME	reasonable maximum exposure
ROICC	Resident Officer in Charge of Construction
RPM	Remedial Project Manager
RQ	risk question
RRO	residual-range organics
RWQCB	(San Francisco) Regional Water Quality Control Board
SA	skin surface area available for contact
SA_a	skin surface area available for contact, adult
SA_c	skin surface area available for contact, child
SAP	Sampling and Analysis Plan
SFEI	San Francisco Estuary Institute
SFS_{adj}	age adjusted factor
SHSP	Site Health and Safety Plan
SSL	soil screening level
SSPORTS	Supervisor of Shipbuilding, Conversion and Repair, Portsmouth
S.U.	standard units for pH
SUF	site use factor
SVOC	semivolatile organic compound
SWAT	solid waste assessment test
SWBZ	second water-bearing zone

SWDIV	Southwest Division Naval Facilities Engineering Command
SWRCB	(California) State Water Resources Control Board
t	time, equal to exposure duration
T	exposure interval
t*	time to reach steady state
t_{event}	event duration
TBA	*tert*-butyl alcohol
TBD	to be determined
TBT	tributyltin
TCA	trichloroethane
TCDD	2,3,7,8-tetrachlorodibenzo-*p*-dioxin
TCE	trichloroethene
TCRA	Time Critical Removal Action
TD	total depth
TDEM	time-domain electromagnetics
TEF	toxicity equivalency factor
TEL	threshold effect level
TEM	transient electromagnetic
TEQ	toxicity equivalent
TIE	toxicity identification evaluation
TOC	total organic carbon
TPH	total petroleum hydrocarbons
TRV	toxicity reference value
TSCA	Toxic Substances Control Act
TtEMI	Tetra Tech EM, Inc.
TTFW	Tetra Tech Foster Wheeler
U	upland
U_m	mean annual wind speed
U_t	equivalent threshold value of the windspeed at 7m
UCL	upper confidence limit
$UF_{soil\ to\ prey}$	uptake factor between soil and prey
URF	unit risk factor
USACE	United States Army Corps of Engineers
USDA	United States Department of Agriculture
U.S. EPA	United States Environmental Protection Agency
USFWS	United States Fish and Wildlife Service
UTL	upper threshold limit
V	fraction of vegetative cover
VF	volatilization factor
VKT	fleet vehicle kilometers traveled
VOC	volatile organic compound
W	wetland
WET	wetland evaluation technique
WHO	World Health Organization

ρ_b	dry soil bulk density
θ_a	air-filled soil porosity
θ_w	water-filled soil porosity
α	attenuation factor
τ_{event}	lag time per event
λ	decay constant

1.0: INTRODUCTION

This Draft Remedial Investigation (RI) Report has been prepared for Naval Facilities Engineering Command, Southwest (NAVFAC SW) under Contract No. N68711-01-D-6009, Task Order No. 0007, as part of the RI/Feasibility Study (FS) evaluation for the West Beach Landfill and Wetlands located at the former Naval Air Station (NAS) Alameda in Alameda, California. The purpose of this RI Report is to document characterization methods employed and the data generated, the data interpretation and evaluation completed, and the conclusions regarding the nature and extent of contamination and the ecological and human health risk posed by contamination at the West Beach Landfill and Wetlands. The RI is being performed under the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) to support cleanup and the ultimate transfer of the West Beach Landfill and Wetlands for future reuse.

The West Beach Landfill and Wetlands are located in the far southwestern corner of Alameda Point and are collectively identified as Installation Restoration (IR) Site 2, which coincides with CERCLA Operable Unit (OU) 4A. The former NAS Alameda is currently referred to as Alameda Point. Throughout this document, IR Site 2 is also referred to simply as "the site". This RI Report evaluates historical data generated by various investigators and more recent data generated during the Navy's implementation of the RI Sampling Work Plan (Battelle et al., 2005), summarizes the nature and extent of contamination present in various environmental media, assesses risks to human health and ecological receptors, and identifies media and/or locations at IR Site 2 that should be evaluated through the FS. The methodology followed during the RI to characterize the nature and extent of contamination at the site and to conduct human health and ecological risk assessments is described in detail in this document.

1.1 Objectives

The primary objectives of this RI Report are to characterize the environmental conditions at the site, describe the nature and extent of contamination in various environmental media, identify areas of potentially unacceptable ecological or human health risk, and define any areas requiring evaluation through an FS of remedial alternatives. The specific objectives are as follows:

- Describe the physical site conditions and ecological setting;

- Describe the methodology followed during the RI evaluation to characterize the nature and extent of contamination and to conduct risk assessments for human and ecological receptors at the site;

- Describe the nature and extent of contamination in various environmental media at the site based on the distribution of chemical constituents identified through characterization activities; and

- Present the results of the ecological and human health risk assessments and determine if there are potential unacceptable risks to human health and/or the environment that would require evaluation in an FS.

1.2 General Approach

The general methods and approach relied on in developing this RI Report are described below.

1.2.1 Nature and Extent of Contamination

During multiple historical and more recent characterization events at IR Site 2, numerous samples have been collected of various environmental media (e.g., soil, groundwater, sediment, surface water, or plant/animal tissue) and analyzed for extensive suites of chemical constituents. All suitable and available data from these various historical and recent characterization events were compiled and evaluated to determine the types of chemical constituents present in environmental media at IR Site 2 and the magnitude and pattern of their occurrence. Laboratory data were evaluated in detail, and conclusions were developed regarding the principal contaminants and the distribution of these contaminants at the site. Analytical chemistry data were compared to relevant and appropriate screening criteria to determine what constituents in various media represent contamination. Background conditions also were assessed by evaluating scientific literature containing documented sampling results from environmental media in the San Francisco Bay area, and by evaluating sampling data generated in conjunction with a reference station sampling event implemented in support of this RI. Assessing background conditions allowed a determination of the types, magnitude, and extent of chemical constituents present ubiquitously in the regional environment versus the types, magnitude, and extent of chemical constituents present at IR Site 2 due to historical operations at the former NAS. Section 5.0 of this report summarizes the evaluation of the nature and extent of contamination at IR Site 2.

1.2.2 Human Health Risk Assessment

The purpose of a human health risk assessment (HHRA) is to evaluate the potential for adverse human health effects from exposure to contaminants at a site. Following guidance from the United States Environmental Proection Agency (U.S. EPA) (1989a), the United States Navy (Navy) (2001), and the California Department of Toxic Substances Control (DTSC) (1996a), the HHRA for IR Site 2 was performed using a tiered process. The first tier was a screening-level assessment that included the development of a preliminary conceptual site model (CSM) focusing on pathways between site contamination sources and humans, identification of contaminants of potential concern, and a screening-level assessment of contaminant exposure and human health effects. The second tier was a baseline assessment that formulated specific exposure scenarios, defined specific exposure assumptions, and more thoroughly screened contaminants of potential concern. In the baseline assessment, exposures and effects were modeled for a number of specific endpoints determined to most conservatively represent the potential risk to humans, and human health risks were quantified for each of these endpoints. Section 6.0 of this report summarizes the HHRA completed for IR Site 2.

1.2.3 Ecological Risk Assessment

The purpose of an ecological risk assessment (ERA) is to evaluate the potential for adverse effects to the natural environment and ecological receptors through exposure to contaminants at a site. As with the HHRA, and in accordance with guidance from U.S. EPA (1997d) and the U.S. Navy (1999), the ecological risk assessment for IR Site 2 was performed using a tiered process. The first tier was a screening-level assessment that included the development of a preliminary CSM focusing on pathways between site contamination sources and ecological receptors and the natural environment, identification of contaminants of potential ecological concern, and a screening-level dose response assessment. The second tier was a baseline assessment that formulated specific exposure scenarios, defined specific exposure assumptions, and more thoroughly screened contaminants of potential ecological concern. In the baseline assessment, exposures and effects were modeled for a number of specific endpoints, and ecological risks were quantified for each of these endpoints. Section 7.0 of this report summarizes the ecological risk assessment completed for IR Site 2.

1.3 Report Organization

This Draft RI Report is organized as follows:

Section 1.0: Introduction. Presents the specific objectives and general approach of this RI Report.

Section 2.0: Site Setting and Description. Provides a description of the site, including location, various physical features, geology, hydrology, and demography, describes the ecological setting at the site, and summarizes the intended future site use.

Section 3.0: Previous Environmental Investigations. Summarizes historical and more recent environmental investigations that have been conducted at IR Site 2 to generate the data used in developing this RI Report.

Section 4.0: Remedial Investigation Methodology and Data Collection. Describes the methodology used in evaluating the nature and extent of contamination and ecological and human health risks at IR Site 2, and describes in detail the characterization activities implemented at the site in accordance with the RI Sampling Work Plan (Battelle et al., 2005).

Section 5.0: Nature and Extent of Contamination. Describes the presence and distribution of contamination in various environmental media at the site based on the evaluation of data generated through historic and more recent characterization events.

Section 6.0: Human Health Risk Assessment. Summarizes the various stages and provides the conclusions of the HHRA.

Section 7.0: Ecological Risk Assessment. Summarizes the various stages and provides the conclusions of the ERA.

Section 8.0: Potential Risk Drivers and Contaminant Fate and Transport. Describes the potential risk drivers identified in the HHRA and ERA and the sources, mechanisms of transport, and exposure scenarios related to those potential risk drivers.

Section 9.0: Summary and Conclusions. Provides a synopsis of the RI Report, including the assessment of the nature and extent of contamination and the risk assessments, and defines the next steps that will occur following the finalization of this RI Report.

Section 10.0: References.

Appendix A: Aerial Photography Review

Appendix B: Field Summary Report for Remedial Investigation Field Sampling Activities

Appendix C: Technical Summary of Toxicity and Bioaccumulation Testing

Appendix D: Analytical Data for Remedial Investigation Field Sampling Activities

Appendix E: Plan View Chemical Concentration Maps

Appendix F: Bubble Plots Representing Chemical Concentrations in Environmental Media

Appendix G: Data Management, Statistical Calculation Procedures, and Sample Chains of Custody

Appendix H: Box and Whisker Plots Comparing Chemical Concentrations in Environmental Media

Appendix I: Wetland Pond Water Quality Monitoring Data

Appendix J: Human Health Risk Assessment

Appendix K: Ecological Risk Assessment

Appendix L: Historical Boring Logs

Appendix M: Responses to Regulatory Comments on the Draft Remedial Investigation Report dated December 8, 2005

2.0: SITE SETTING AND DESCRIPTION

This section summarizes the location and history of IR Site 2, as well as the general physical, climatologic, geologic, hydrologic, and ecological characteristics of the site. This section also summarizes the currently proposed plan for transfer and future use of IR Site 2.

2.1 General Location and Site Description

Alameda Point is located on the western end of Alameda Island, which lies on the eastern side of San Francisco Bay, adjacent to the City of Oakland. The locations of Alameda Point and IR Site 2 are depicted on Figure 2-1. Overall, Alameda Point encompasses roughly 1,700 acres of land. Development of Alameda Point first began in 1930 under the ownership of the United States Army (Army), and the majority of the former NAS was built on shallow open water through dredging and filling. The average elevation of Alameda Point is only 15 feet (ft) above mean sea level.

Alameda Point served as a base of operations for Naval surface craft from prior to World War II until its closure in 1997. Closure of Alameda Point was mandated by the Defense Base Realignment and Closure Act (BRAC) of 1990. During its long history of operations, Alameda Point was home to several thousand military and civilian personnel, and supported operations of the Navy, Marine Corps, and other military entities. Hundreds of buildings and an extensive network of roadways and utilities were constructed at Alameda Point, and much of this infrastructure still exists. Alameda Point supported aviation activities through extensive runway and tarmac infrastructure and an enclosed lagoon for seaplanes, and also supported naval surface vessels (including aircraft carriers) through an extensive system of piers, berthing areas, and turning basins. Specific activities conducted historically at Alameda Point included, but are not limited to, aircraft maintenance, ship maintenance, support and training for Naval and Marine air units, storage, rework, and distribution of weaponry, fuel storage and refueling, dry goods storage and distribution, pest control, plating, metal working and fabrication, parts washing, cleaning and routine maintenance, blasting and painting, testing of jet engines, heavy equipment maintenance, woodworking, photography, and radiological operations that included the painting of aircraft dials with radioluminescent paints.

IR Site 2 is located on the southwestern corner of Alameda Point. The general layout of IR Site 2 is shown on Figure 2-2. The site consists of the West Beach Landfill (herein also referred to simply as the landfill), which occupies approximately 77 acres, and the West Beach Wetlands (herein also referred to simply as the wetlands), which covers approximately 33 acres immediately south and west of the landfill. The site is bounded to the south and west by San Francisco Bay and to the east and north by runways, tarmacs, and related features. The landfill was reportedly used for disposal of waste generated by former NAS Alameda activities from 1956 through early 1978.

2.2 Site History

The area of present day IR Site 2 was originally shallow open water. In 1956, a sea wall was constructed along the southern and western shorelines of IR Site 2 to confine and protect the area (see Figure 2-2). Dredged fill of varying origins was hydraulically placed inside the sea wall, essentially creating IR Site 2. Observations made during the current RI indicate that the sea wall surrounding IR Site 2 remains intact and that its integrity has not been compromised.

Although historical information suggests that limited waste disposal activities may have occurred in the far northeastern portion of the site during the early 1950s, disposal activities began in earnest at the landfill only after completion of the sea wall in 1956. Waste was initially placed starting in the northern portion of the landfill, eventually extending to most of the northern and eastern areas of the landfill and part of the northern wetland pond. It is estimated that the landfill received a maximum of 1.6 million tons of

general base garbage (i.e., general household waste, food waste, paper products, etc.) over its operation as the main disposal location for Alameda Point. In addition, the following specific wastes were reportedly disposed of at the landfill based on an Initial Assessment Study (IAS) historically commissioned by the Navy (Ecology and Environment, Inc. [E&E], 1983): waste chemical drums; solvents; oily waste and sludge; paint waste; plating wastes; industrial strippers and cleaners; acids; mercury; polychlorinated biphenyl (PCB)-containing liquids; batteries; low-level radiological waste from radium dials and dial paints; scrap metal; inert ordnance; asbestos; several pesticides (solid and liquid); tear gas agent; biological waste (i.e., potentially infectious waste from Oak Noll Naval Hospital and laboratory waste from the Naval Supply Center at Oakland); creosote; dredge spoils; and waste medicines and reagents. Specific disposal activities that were identified through an evaluation of facility records and/or interviews with former NAS Alameda personnel during the IAS include:

- Asbestos pipe lagging removed from ships was placed in central and eastern parts of the landfill;

- Pesticides were disposed of in the southeastern portion of the landfill;

- Roads on the landfill were constructed in part from spent sandblasting grit and abrasives used to rework metal parts and strip paint from ship hulls;

- Waste oils and PCB-containing oils were used as dust suppressants on the roads in the northeastern corner of the landfill;

- Several hundred pounds of tear gas agents were disposed in containers as a loose powder at the landfill in 1968 or 1969;

- Mercury waste from manometers and fluorescent light bulbs was drummed and disposed at the landfill between 1952 and 1968;

- Two unlined pits in the landfill (one located in the northeastern corner of the landfill and one in the mid-western portion of the site, immediately south of the landfill/wetland boundary) were used to contain waste oils; and

- Sludge and grit from 15 oil-water separators located at Alameda Point were disposed of at a rate of approximately 53,000 gallons of sludge per month during the 1960s.

Historical disposal methods at the site generally consisted of trench and fill operations. A trench would be excavated to the water table and progressively filled with wastes, then the waste material would be spread and compacted using heavy machinery. The landfill area was covered with cover soil on an intermittent basis. Reportedly, during its early years of operation, full drums were buried at the site. However, after three separate landfill fires were triggered by damaging full drums of material with heavy machinery, all drums were reportedly punctured and drained before being disposed (E&E, 1983).

After landfill operations ceased in 1978, an earthen berm was constructed around the perimeter of the site, completely physically containing the site. This berm is an earthen surface feature, and likely does not in and of itself act to contain groundwater, leachate, or landfill gas. The landfill is currently defined as extending to the inside edge of this berm or to the edge of the wetland area (see Figure 2-2). The wetland area is located in the southwestern corner of Alameda Point and contains two perennial surface water ponds. Due to their location within the wetland portion of the site, these ponds are referred to as wetland ponds (see Sections 2.10.2 and 2.10.3). The northern wetland pond (also known as the North Pond) is hydrologically connected to the San Francisco Bay through a 36-inch (in)-diameter pipe culvert that

penetrates the perimeter berm and seawall, and contains water year round (see Figure 2-2). General facility waste reportedly was placed in at least a portion (the northern margins) of the North Pond during the course of landfill operations, and subsequently covered with fill soils. The southern wetland pond (also known as the South Pond) was created by excavating soil to the water table for use as landfill cover, and subsequently was filled with fresh water via precipitation. Given its lack of connection to San Francisco Bay, the inundated extent of the South Pond is seasonally variable and water is replenished primarily during precipitation and runoff events. Although the South Pond was not routinely used in any way for waste disposal during the operation of the landfill, it is reported that scrap metals from the Public Works Center may have been disposed in the South Pond during the 1970s and that 24,000 cubic yards (yd^3) of dredged material from the southern side of Seaplane Lagoon (i.e., IR Site 17/OU 4B at Alameda Point) was deposited in this area (E&E, 1983).

Two areas within IR Site 2 are identified as the coastal and interior margins. The coastal margin is the area between the landfill or wetlands and San Francisco Bay. This strip of land buffers the landfill and wetlands from San Francisco Bay and consists of the perimeter berm and sea wall. The interior margin is the area of the site that bounds the landfill on the north and east. This region contains sections of the perimeter berm and includes all areas outside the berm to the north and east that are part of IR Site 2. Both margin areas are defined solely for the purpose of identifying the locations of samples collected during the historical investigations, and are shown on Figure 2-2.

In 1978, the Navy developed plans to close the landfill in accordance with the requirements of the San Francisco Regional Water Quality Control Board (RWQCB) Minimum Criteria for Proper Closure of Class II Solid Waste Disposal Sites (Resolution No. 77-7). In 1983, the RWQCB issued Order No. 83-35 to implement final cover, a leachate cutoff barrier, methane gas control, earthquake damage control, drainage control, and erosion control, and to generate compliance reports for the landfill. The Navy complied between 1983 and 1985 by placing a partial clay-soil cover, installing an 820-ft-long, 2-ft-wide, 20- to 30-ft-deep slurry wall to restrict potential contaminant migration to San Francisco Bay, installing a gas venting system, and completing repairs to the sea wall. Closure activities were discontinued in 1984 because the RWQCB required that the Navy first complete a solid waste assessment test (SWAT). In 1986, the Navy spread 20,000 yd^3 of imported soil material on the landfill. That volume represented a shortfall of 55,000 yd^3 to achieve a uniform cover layer of appropriate thickness over the landfill area. Also in 1986, the landfill was graded to eliminate depressions that could yield ponding during precipitation events, and an earthen perimeter levee was constructed around the landfill.

In August 1999, IR Site 2 (along with other sites at the former NAS) was officially added to U.S. EPA's National Priority List (NPL) of Superfund sites and assigned Comprehensive Environmental Reponse, Compensation, and Liability Information System (CERCLIS) identification CA2170023236.

Aerial photography showing Alameda Point is available for certain years between 1930 and 2003. Site conditions visible in the photographs are consistent with historical site construction information and information describing historical waste disposal practices. A detailed aerial photograph review is provided in Appendix A of this RI Report.

2.3 Demographics and Land Use

IR Site 2 was used as a landfill between the mid 1950s and latter 1970s. There was no land use at the site prior to 1956, when the perimeter seawall was constructed and the site was first built using dredged fill. Due to its sole historical use as a landfill, no persons have resided or currently reside at the site. Because it has never been used for full time residence or occupancy, no above- or belowground utilities (e.g., potable water, electric, or telephone) exist at the site.

2.4 Surface Features

Overall, the site can be divided into the following five general areas:

- Upland areas consisting predominantly of the landfill;

- Wetland areas south and west of the landfill;

- The North Pond and South Pond located in the wetland areas;

- The coastal margin comprised of the perimeter rock sea wall and berm along the western and southern boundaries of the site; and

- The interior margin consisting of the berm north and east of the landfill and all areas north and east of the berm that are part of the site.

IR Site 2 topography gently slopes overall from the north and east toward the south and west, in the direction of the San Francisco Bay. The topography of the landfill is graded to promote surface drainage and consists predominantly of upland habitat. Visual observations of the site do not suggest the presence of significant erosion channels or areas that retain standing water following precipitation. The upland area is covered predominantly with grasses, shrubs, trees, and roads that traverse the landfill. Miscellaneous landfill waste and debris has become exposed through the landfill soil cover over the course of time, or may not have been properly covered during historical closure activities. The wetlands are covered by salt-tolerant wetland plant species and pockets of grasses, shrubs, and small trees.

The topography of the wetland and upland habitat is locally irregular. Small isolated areas occur within the wetlands that would more appropriately be considered upland. In addition, isolated areas within the upland portion of the site historically have been mapped as transient wetlands that support hydrophytic vegetation and/or in which soils remain wetted for a significant portion of the year (see Figure 2-2). The persistent emergent wetlands at the site have been mapped as approximately 29.3 acres in extent, and four individual areas more appropriately considered wetland have been mapped in the northeastern portion of the site. The aggregate area of the four isolated wetland areas in the northeastern portion of the site is approximately 4.5 acres (Tetra Tech Foster Wheeler, Inc. [TTFW], 2004). A large earthen berm forms the perimeters of the wetland and landfill, and an unpaved road is present along the spine of the berm. A sea wall constructed of large boulders borders the southern and western boundaries of the site. An isolated beach area of approximately 200 ft long 20 to 30 ft wide has been observed only at low tide along the San Francisco Bay at the northwestern boundary of the site. The beach area and associated sampling that has been performed as part of the Offshore Sediment Study (Battelle et al., 2005) are discussed in greater detail in Section 6.0 of this RI Report. The North and South Ponds in the wetlands are the only two isolated areas of year-round surface water ponding at the site. However, the surface area of each pond fluctuates seasonally and flooding has been observed in areas of the wetlands besides the ponds. A 36-in-diameter corrugated steel pipe culvert at the western side of the wetlands penetrates the perimeter berm and seawall and connects the North Pond to San Francisco Bay. Otherwise, the berms and seawall generally prevent direct surface water exchange between the wetlands and San Francisco Bay.

The only permanent structures that are documented as having existed at the site include a radioactive waste storage shack formerly located in the far northwestern portion of the site and two earthen ammunition bunkers in the northern portion of the site, all within the interior margin at Site 2 (see Figure 2-2). The radioactive waste storage shack was historically demolished, but the earthen bunkers are still present. A network of landfill gas vents is present at IR Site 2. In addition, numerous groundwater and gas monitoring wells are present across the surface of site, primarily along the berms. In total,

42 permanent monitoring wells or well clusters and five gas monitoring well clusters are present at IR Site 2. The portions of the site bounded by land (i.e., the north and eastern perimeters) are completely secured with metal chain-link fencing. Physical access to the site is controlled through a single locked gate at the far northwestern corner of the site. Although the western and southern perimeters of the site are not fenced, the rocky sea wall creates highly difficult access conditions and there is signage at regular intervals along the shoreline indicating the potential hazard associated with the site.

2.5 Surface Soils

The surface soil across Alameda Point is described by the United States Department of Agriculture (USDA) Natural Resources Conservation Service as Xeropsamments, defined as a very permeable, sandy fill material dredged from old beach areas (USDA, 1981). This classification is consistent with historical site construction information, which indicates that IR Site 2 was originally constructed of dredged material taken from the shallow water environment and that the landfill was covered with similar material to that used to originally construct the site. Typically slow surface runoff is associated with the Xeropsamment soil type, and the potential for surface water erosion is generally low. Based on RI field sampling activities (described in Sections 3.0 and 4.0 of this RI Report), the majority of the surface soil at IR Site 2 is recent, nonweathered, mineralized sand with low clay content. Also, based on field observations during wetland delineation efforts in 2004 (TTFW, 2004), the surface soils in the wetland area(s) meet the 1991 National Technical Committee for Hydric Soils (NTCHS) indicator criteria for hydric soils (USDA Soil Conservation Service [SCS], 1991).

2.6 Meteorology and Climate

Alameda Point has a maritime climate which consists of mild summers and winters. The majority of the precipitation that occurs in the San Francisco Bay area occurs between October and April. The average annual rainfall in the San Francisco Bay area is approximately 20 to 22 in (NOAA, 2005). At the site, precipitation either returns to the atmosphere via evapotranspiration, runs off into the ponds or into San Francisco Bay, enters San Francisco Bay through exchange with the North Pond, or infiltrates into the subsurface and groundwater underlying the site.

The daily high temperature generally averages 50 degrees Fahrenheit (°F) from November through April and 80°F from May through October. The median daily high temperature is approximately 65°F. Freezing temperatures are generally never experienced, and frozen precipitation is exceedingly rare. Based on measurements from a monitoring station in Alameda, San Francisco Bay surface water temperatures are on average in the low 50°F range during the winter months and the mid 60°F range during the summer months. The dominant wind direction in the Oakland area is from the west or west northwest, with average annual wind speeds of approximately 7 to 7.5 miles per hour (mph). Wind speeds typically are greatest in the late spring and summer months, and lowest in winter months.

2.7 Surface Water Hydrology

The site is bounded to the west and south by San Francisco Bay. The surface water features present at the site consist of two wetland ponds (the North Pond and South Pond). These ponds receive direct recharge from precipitation. In addition, the ponds are recharged by storm water runoff within their respective local watersheds during and following rainfall events. The watersheds of the ponds include the landfill and wetland areas and the steep inner slope of the perimeter berm. The North Pond is in communication with the San Francisco Bay through a pipe culvert that penetrates the perimeter berm and sea wall, which would allow more saline bay water to enter the pond and fresh water to exit. As such, geochemical characteristics in the North Pond are largely controlled by precipitation, evaporation, and exchange with San Francisco Bay, whereas geochemical properties in the South Pond are largely controlled by precipi-

tation and evaporation. Given the shallow depth to groundwater in the wetland area of the site, it is likely that the surface water in the ponds also is in communication with shallow groundwater beneath the site during at least certain times of the year. This relationship between surface water and groundwater is discussed in more detail in Section 2.8.2 below.

2.7.1 North Pond

The North Pond covers approximately 9.2 acres and holds water throughout the year. A pipe culvert directly connects the North Pond to San Francisco Bay. A historic SOMA Corporation hydrologic study (2000) documented that water level measurements indicate a consistent downward vertical hydraulic gradient in the North Pond, suggesting the pond is a consistent source of recharge to groundwater. The study also found that the North Pond consistently discharged to the San Francisco Bay through the pipe culvert during the wet and dry seasons, except during the high tide portion of the tidal cycle. Salinity measurements collected in the North Pond are lower in the wet months, indicating the rate of freshwater recharge via precipitation exceeds the rate of recharge from the more saline water in San Francisco Bay. Measured salinity generally increases in the dry months, indicating the rate of saline water recharge from San Francisco Bay is greater than recharge from any freshwater source. Given its connection to San Francisco Bay, depth of surface water in the North Pond is more variable and tends to be muted even during periods of significant precipitation.

2.7.2 South Pond

The South Pond spans 6.3 acres and also generally holds standing water throughout the year. However, due to its lack of connectivity with San Francisco Bay, the South Pond is more prone to drying and the surface area of this pond is significantly less during the dry season compared to the wet season. The South Pond is not hydrologically connected to the North Pond, but during the wet season it is conceivable that the two ponds could be interconnected if significant precipitation would cause either pond to overflow its banks into the surrounding wetland area. Measured salinity in the South Pond decreases during the wet season in response to freshwater input from precipitation and increases in the dry season, likely because salts are concentrated in the South Pond through the precipitation/runoff/evaporation cycle. Surface water in the South Pond typically reaches greater depths than the North Pond because water depth is not muted by a connection to San Francisco Bay.

2.7.3 San Francisco Bay

San Francisco Bay bounds IR Site 2 on its western and southern perimeters. The bay is in communication with the North Pond through a pipe culvert that penetrates the sea wall and perimeter berm. In addition, a recent tidal study (Shaw Environmental Inc. [Shaw], 2005) found that tidal effects exist in shallow groundwater in some areas adjacent to the San Francisco Bay with tidal efficiencies greater than 8 percent (%). These results indicate shallow groundwater adjacent to the bay is influenced by San Francisco Bay surface water, and, in turn, it is possible that shallow groundwater at the site is in communication with surface water in the wetland ponds. However, surface water in the ponds was not included in the recent tidal influence study (Shaw, 2005).

2.8 Geology and Hydrogeology

2.8.1 Geology

The geology of IR Site 2 has been well characterized through the historic installation of 42 monitoring wells, 31 geotechnical borings, and 7 cone penetrometer testing (CPT) lithologic soundings and more recently through numerous borings during the current RI. Site geology is generally consistent with the

geology found in other portions of Alameda Point. The unconsolidated lithologic units present at IR Site 2 include (from oldest to youngest and deepest to most shallow) the Alameda Formation, the lower and upper units of the San Antonio Formation, the Merritt Sand, the Bay Sediment Unit (BSU) and a subunit of the the BSU known as the Young Bay Mud, and artificial fill. A clay known as the Yerba Buena Mud comprises the lower unit of the San Antonio Formation, and is considered to be an effective hydraulic barrier (i.e., functions as an effective aquiclude/aquitard) between the Alameda formation and overlying strata. The Yerba Buena Mud (also known as the Old Bay Mud) consists of firm, gray, silty clay and is continuous across the site. The Yerba Buena Mud reportedly occurs at depths between approximately 80 ft below ground surface (bgs) in the southern portion of the site to approximately 110 ft bgs in the northern portion of the site. The upper unit of the San Antonio Formation generally underlies the Merritt Sand and consists of interbedded layers of sand and clay with a thickness of approximately 10 ft.

The Merritt Sand is composed of brown, fine- to medium-grained, poorly graded sand and underlies the Holocene-age deposits of the BSU and Young Bay Mud. At IR Site 2, the Merritt Sand is approximately 25 ft thick and occurs primarily in the southern portion of the site. The Young Bay Mud consists of silt and gray to black clay, and is generally interbedded with the BSU, which consists of poorly graded, silty and clayey sand layers. The Young Bay Mud is generally continuous across IR Site 2, though it does appear to thin out or become locally discontinuous in the eastern and southern portions of the site. This subunit is at least 40 ft thick in the northwestern portion of IR Site 2. The BSU is up to 50 ft thick and is generally site-continuous with the exception of the southeastern corner of the site, where it appears to be locally absent. Given their generally fine-grained lithology, the BSU (including the subunit Young Bay Mud) generally functions as an aquiclude/aquitard between two discrete shallow aquifer water bearing zones across most of the site (see Section 2.8.2 below). The artificial fill at the site consists of dredged material used originally to construct IR Site 2, waste material deposited during the operation of the site as a landfill, and imported material used to construct the site perimeter berm. The thickness of this artificial fill layer is highly variable across the site, but is generally 20 to 30 ft.

Figure 2-3 shows a plan view of various cross sections depicting the geology of IR Site 2. Figures 2-4 through 2-9 present the individual geologic cross sections, and are based on the numerous historical subsurface borings completed across the site. Copies of geologic logs for subsurface borings that have been historically advanced at the site, including some used to compile the cross sections in Figures 2-4 through 2-9, are provided in Appendix L of this RI Report.

On a more regional scale, the San Francisco Bay region is bounded by the Santa Cruz Mountains to the southwest and the East Bay Hills and Diablo Range to the northeast. The nearest active fault to the site is the Hayward Fault, which is approximately 7 miles east of the site. The San Andreas Fault is located within the hills on the west side of San Francisco Bay approximately 12 miles from the site. Other major faults in the region include the Calaveras Fault system and the Green Valley and Greenville Fault systems, which are all on the east side of the East Bay Hills.

2.8.2   Hydrogeology

Alameda Point is underlain by two distinct groundwater aquifers. The deep aquifer occurs in the Alameda Formation, and the shallow aquifer occurs above the Yerba Buena Mud aquitard/aquiclude. Beneath IR Site 2, the shallow aquifer is composed of a first and second water bearing zone. The first water-bearing zone (FWBZ) beneath IR Site 2 is an unconfined (i.e., water table) aquifer generally occurring in the artificial fill material, with depth to groundwater ranging from 2 to 8 ft bgs and averaging 3 to 5 ft bgs. The saturated thickness of the FWBZ is up to approximately 30 ft in the western portion of the site (see Figure 2-9). Groundwater in this water bearing zone is typified by shallow gradients and low hydraulic conductivities, indicating relatively low groundwater velocities. Therefore, groundwater

discharge rates from this zone are likely to be relatively low. Based on tidal studies performed at the site, the FWBZ is influenced by tidal fluctuations of San Francisco Bay (Shaw, 2005).

The second water-bearing zone (SWBZ) is semiconfined and generally occurs in the Merritt Sand and/or the upper unit of the San Antonio Formation where these units exist, and/or in the lower portion of the BSU where this unit is comprised of coarser-grained material. Where the lower BSU forms the SWBZ, the Merritt Sand and upper unit of the San Antonio Formation are generally absent. The SWBZ is confined from the FWBZ by the Young Bay Mud/BSU aquitard/aquiclude, which is regionally discontinuous but generally continuous across the site and is generally sufficiently fine-grained to be considered a zone incapable of transmitting and/or yielding groundwater. The SWBZ varies in thickness from 0 to approximately 50 ft , with greater thicknesses appearing to result from historical erosion associated with a northeast-to-west-trending paleochannel at Alameda Point. There is no connection between the shallow aquifer systems on Alameda Point and the Oakland mainland because Oakland Inner Harbor bisects the Merritt Sand unit. The Merritt Sand unit on Alameda Point is therefore hydrologically isolated from mainland aquifers.

Based on water level measurements obtained throughout the monitoring well network at the site, groundwater flow in the FWBZ at IR Site 2 is generally in the direction of San Francisco Bay. In the northern and central portions of the site, groundwater flow is predominantly towards the west. However, areas in the northern and central portions of the site exhibit shallow groundwater flow in locally variable directions. In the southern portion of the site, FWBZ groundwater flow is typically towards the south or southwest in the direction of the coastal margin and San Francisco Bay. In the SWBZ, groundwater flow is predominantly towards the west and southwest. Figure 2-10 shows groundwater flow in the FWBZ and SWBZ at IR Site 2 based on water level measurements generated through an ongoing quarterly groundwater monitoring program (see Sections 4.0 and 5.0 of this RI Report for more detail on the ongoing quarterly monitoring program) in 2004.

Groundwater elevations observed in monitoring wells located within the landfill have been generally higher than those measured in the wetland and surface water elevations observed in the northern and southern wetland ponds. These observations suggest that groundwater beneath the landfill is potentially a source of hydrologic recharge to the wetland ponds and wetland groundwater. Given its proximity to San Francisco Bay, groundwater flow conditions at IR Site 2 are influenced to some extent by tidal fluctuations. For instance, Figure 2-10 appears to demonstrate local FWBZ groundwater flow in the western portion of the site towards the east, which could be related to the tidal conditions at the time those groundwater elevation data were collected. In addition, at least the northern wetland pond is in communication with San Francisco Bay through the pipe culvert that penetrates the perimeter berm and seawall. In general, the maximum influence of tides on groundwater flow conditions is observed along the southern and western edges of IR Site 2, immediately adjacent to the bay (Shaw, 2005).

2.9 Landfill

The landfill occupies the northern and eastern portions of the site and covers approximately 77 acres. As described above, over its history, the landfill is estimated to have accepted approximately 1.6 million tons of general facility waste and several discrete and potentially hazardous wastes. The landfill was not con-structed with a liner, and the most common disposal method throughout the course of the landfill's opera-tion was trench and fill. The total thickness of waste material in the landfill corresponds approximately to the depth to groundwater, which is approximately 6 to 10 ft bgs across the landfill. Given variability in the groundwater surface and the fact that historical waste trenches likely extended at least some depth below groundwater, some amount of landfill waste is likely saturated at any given time. The thickness of soil cover on the landfill varies from approximately 2 in to 2 ft. The existing landfill soil cover is inconsistent, poorly compacted, and highly permeable. Grain size of the cover soil is generally medium

to fine sand with some amount of silt and gravel, and the soil is generally loose. Waste material can be seen at the surface in some locations across the landfill. Waste material present at the ground surface and encountered during invasive investigation activities includes items such as wire, asphalt, wood, plastic, rubber, and metal. Landfill fill material also contains gravel, shell fragments, wood chips, and concrete rubble. No significant odors are noticeable at the landfill and no odors have generally been detected during invasive activities. Soil discoloration is commonly observed in subsurface borings and test pits. The fill is generally poorly compacted and blow counts measured during the completion of invasive borings using a drill rig and/or Geoprobe® have generally been in the low single digits (i.e., less than 5). Because of these conditions, the existing landfill soil cover would likely be unsuitable for use as part of any final cover design.

The BSU and the Young Bay Mud, a fine-grained, cohesive aquiclude/aquitard subunit of the BSU, are present below the fill material and extend to depths up to 70 ft bgs. Blow counts recorded in the Young Bay Mud zone have been relatively low. The BSU is present as a light olive brown sand layer with significant fines and is relatively dense with recorded blow counts consistently reaching 50 or even refusal.

As indicated above, a slurry wall was constructed at the site in the mid 1980s to prevent the potential for contaminated groundwater to enter San Francisco Bay. The slurry wall is present along the western edge of the landfill and is approximately 820 ft long, 2 ft wide, and 20 to 30 ft deep. The slurry wall does not extend into the wetland portion of the site.

Historically, Foster Wheeler Environmental Corporation (FWEC) prepared a *Draft Final Ordnance and Explosives Waste/Geotechnical Characterization Report* (FWEC, 2003) which included some characterization of the landfill at IR Site 2. As part of that characterization, a geotechnical and seismic assessment was conducted for the site. The geotechnical evaluation, which included the assessment of ground settlement from hypothetical surface loads, was conducted across IR Site 2 and in the immediate offshore area. The maximum ground settlement expected to occur from a hypothetical 4-ft-thick landfill cap was determined to be approximately 13 in. Ground settlements would be expected to occur over a long period of time (i.e., 40 years or more), and thus they would not pose an immediate hazard. Also, as part of its geotechnical assessment, FWEC analyzed different cross sections at the site for static (pre-earthquake) stability. The PC-STABL-5M program, which is based on limit equilibrium theory, was used to obtain a factor of safety against slope failure at the site (Achilleos, 1988). This factor is defined as the ratio of resisting (stabilizing) forces to the driving forces trying to displace a slope. Guidelines for such stability analyses are provided in Title 27 of the California Code of Regulations (CCR); however, no specific value for the static factor of safety is provided in this regulation. The current state of practice in California at the time the geotechnical analysis was performed was to require a minimum static factor of safety of 1.5. Six different cross sections across IR Site 2 were analyzed with an assumed 4-ft-thick soil cover, and all but one were determined to have a static factor of safety greater than 1.5. The factor of safety one cross section was 1.46 (i.e., marginally below 1.5), which led to the conclusion that some remedial measures involving geotechnical improvements might be needed to increase the static factor of safety to meet the State of California standard of practice.

Potential seismic hazards at IR Site 2 include liquefaction potential and slope instability. Fill material at the site from historical dredging extends to a depth of approximately 20 to 30 ft bgs across the site, with the depth reaching up to 40 ft bgs towards the western and southern edges of the site. Waste material disposed over the course of landfill operation also is a component of the fill material, but comprises only a small fraction of the volume of dredged fill material used to originally construct the site. During the FWEC seismic hazards analysis, an integrated CPT-based method (Robertson and Wride, 1997) was used to quantify the potential for liquefaction and identify areas susceptible to liquefaction. Based on the analyses, the upper fill material at the site was identified as having a high potential for seismic

liquefaction and was designated as liquefiable. Potential liquefaction-induced settlements in the fill layer were estimated to be up to 12 in. The Young Bay Mud underlies the artificial fill at IR Site 2 and could be a relatively compressible and weak layer under seismic stresses. The FWEC seismic analysis determined that settlement in the Young Bay Mud layer could be approximately 4 to 6 in due to liquefaction and consolidation. Total seismically induced settlement of the artificial fill and Young Bay Mud could reach 18 in (FWEC, 2003). Lateral deformation was estimated to be greater than approximately 20 ft. Ground settlements from potential future placement of a landfill cap or additional fill for grading purposes would be anticipated to occur mainly from elastic settlement of the fill layer and consolidation and/or compression of the Young Bay Mud layer. Settlement in the Merritt Sand, San Antonio Formation, and Alameda Formation would likely be negligible because they are very dense and stiff compared to other geologic units. Slope instability also was identified as a major potential seismic hazard. Using Newmark-type procedures, permanent lateral (slope) deformation under seismic stress was predicted to be high, ranging from 4 to 19 ft.

To address the liquefaction potential concerns and other hazards such as seismically induced settlements and lateral displacements, a Geotechnical Feasibility Study Report (FWEC, 2004) previously was developed for IR Site 2. That document screened potentially viable remedial alternatives to address the hazards, and recommended a preferred alternative. The Geotechnical Feasibility Study consisted of outlining the remedial action objective, identifying response actions, developing and screening potential remedial alternatives, developing an implementability analysis, evaluating costs of the various alternatives, and selecting a preferred alternative. The remedial action objective presented in the Geotechnical Feasibility Study was to prevent the release of waste into San Francisco Bay. A total of 20 remedial alternatives were developed in the Geotechnical Feasibility Study by combining various soil improvement and physical buttressing methods. The four most appropriate alternatives were identified using a screening process that considered site-specific conditions and applicable U.S. EPA Feasibility Study screening criteria (i.e., effectiveness, implementability, and cost). An additional and more detailed layer of screening was performed to narrow the list of remedial alternatives to two, namely a soil cement gravity wall with stone columns (i.e., Alternative 5 in the Geotechnical Feasibility Study) and a concrete wall (i.e., Alternative 6 in the Geotechnical Feasibility Study). Based on a detailed and comparative analysis, the soil cement gravity wall with stone columns was determined to be the most feasible and appropriate remedial strategy to mitigate geotechnical/seismic hazards. This approach would involve the construction of a 17- to 38-ft-wide soil cement gravity wall in the Young Bay Mud and the installation of stone columns in the fill layer (from the top of the Young Bay Mud to the ground surface) to reduce liquefaction potential and contain liquefied soils behind the improved soil zone. This alternative was recommended because of the overall safety and reliability of the soil cement gravity wall, and substantial cost efficiency compared to Alternative 6.

The Geotechnical Feasibility Study recommended that during a detailed design stage the remedial measure be closely evaluated to determine if spatial extent and/or other implementation factors can be optimized based on more detailed waste delineation, more sophisticated or detailed analysis (e.g., Finite Element modeling) to obtain an accurate assessment of slope movement, and/or the assessment of risk from the potential for waste release into the San Francisco Bay. In addition, it is possible that the preferred alternative could require refinement or adjustment based on additional data or evaluations performed in the future.

2.10 Ecology

IR Site 2 borders San Francisco Bay and is situated along the Pacific Flyway. Most of the land at Alameda Point was originally created by filling existing tidelands, marshlands and sloughs with dredged material from the bay and Oakland Inner Harbor. The site is generally considered "disturbed" due to historical filling and landfilling activities. The site is ecologically isolated (particularly for mammals that

may use the site) as it is bordered by San Francisco Bay to the south and west, and by roads, runways, and other hardened or impermeable (e.g., concrete) surfaces to the north and east; in addition, the site is fenced along its entire (onshore) boundary. Also, certain higher trophic level species that exist in the site are managed to ensure protection of a least tern colony residing at nearby Alameda runway.

The surface soils and sediments at the site consist largely of coarse, well-drained sands. The underlying soils and sediments are typically finer, consisting of silt and clay and extending up to 120 ft bgs. The site generally consists of three distinct habitat types: upland; non-inundated persistent emergent salt marsh wetlands (i.e., wetlands); and wetland ponds. These habitat types are shown on Figure 2-2. It is important to note that the areas defined as upland, wetland, and wetland pond demonstrate variability in their extent depending on changes in water levels resulting from seasonal rainfall and tidal variability.

In general, the upland portion of IR Site 2 is considered a disturbed environment due to historical filling and landfilling activities. Under current conditions, this area of the site does not support a large diversity or density of wildlife species. The wetland areas support a variety of wetland plant species and avian species that use the site for breeding, foraging, and/or refuge. However, the wetlands themselves do not support a high diversity or density of invertebrates or mammals. Similarly, the wetland ponds may be used by a variety of avian species but do not appear to support aquatic vegetation or significant invertebrate and fish populations.

The three identified habitat types and their associated biota are described in more detail in the following subsections. Table 2-1 summarizes the sources of information used in understanding and describing the ecological setting of IR Site 2.

2.10.1 Upland

The actual landfill portion of IR Site 2 covers approximately 77 acres and, along with the interior margin, constitutes the upland area of the site. The area has been partially covered using fill materials dredged from the wetlands and imported from other sites across Alameda Point (E&E, 1983). The terrestrial environment of the landfill is best characterized as coastal prairie and northern coastal scrub. As discussed previously, the majority of the upland area is highly disturbed from historical landfill activities. As such, the vegetation existing on the upland is currently is a mix of native species such as coyote brush (*Baccharis pilularis*) and seaside trefoil (*Lotus formosissimus*), and non-native species such as ice plant (*Carpobrotus edule* and *C. chilense*) and Italian thistle (*Carduus* sp.), with the latter being dominant. With the exception of a few isolated depressions, these upland terrestrial areas are never or rarely inundated.

Although soil invertebrates have not been specifically surveyed, species historically collected for tissue analysis for the purposes of food web modeling (TetraTech EM, Inc. [TtEMI], 1998a) include various types of beetles, bees, moths, butterflies and spiders. Amphibians and reptiles have not been specifically surveyed, but based on natural history information for the site are expected to occur primarily in the upland areas of the site. Species expected to occur include the Pacific treefrog (*Hyla regilla*), the western fence lizard (*Scelopourus occidentialis*), common garter snake (*Thamnophis sirtalis*), and Pacific gopher snake (*Pituophis melanoleucus catenifer*).

A variety of birds and mammals have been observed at the site during historical surveys (PRC Environmental Management Inc. [PRC], 1995; TtEMI, 1998a). A list of mammalian and avian species observed at the site are presented in Tables 2-2 and 2-3. More than 130 species of birds have been observed at Alameda Point and may use IR Site 2 to some degree. Passerine birds that glean insects off of and feed on terrestrial vegetation are the most significant avian components of the upland terrestrial habitat. However, the most significant bird use is probably associated with the wetlands and wetland ponds, which

provide resting and foraging areas for migrating waterfowl and habitat for other resident bird species. It is noteworthy that no avian foraging from the wetland ponds was observed during the Navy's RI sampling activities of 2004/2005. Commonly observed avian species include the barn swallow (*Hirundo rustica*), European starling (*Sturnus vulgaris*), western meadowlark (*Sturnella neglecta*), and house finch (*Carpodacus mexicanus*). Mammals that potentially live, forage, and/or reproduce in the upland terrestrial habitat include the domestic rabbit (*Oryctolagus cuniculus*), black-tailed hare (*Lepus californicus*), raccoon (*Procyon lotor*), long-tailed weasel (*Mustela frenata*), and striped skunk (*Mephitis mephitis*). The habitat also is suitable for the red fox (*Vulpes vulpes*), but this species has not been specifically observed at the site.

It is important to note that ecological management is currently employed at IR Site 2 for the protection of Caspian and least tern colonies residing at Alameda Point. Potential predators of terns and tern eggs that are managed through this program include red fox, coyote, and carnivorous birds. Specifically, these potential predators are harassed, temporarily relocated, or in extreme circumstances culled periodically by field staff from the United States Fish and Wildlife Service (USFWS) to protect the terns.

2.10.2 Wetlands

Although the roughly 33-acre area within the southwest portion of the site is generally considered wetlands, the actual topography and hydrology within this area is variable. Small islands of upland area occur within the wetlands and small inundated wetland areas occur in depressions within larger non-inundated regions. Two wetland delineation programs were historically conducted by the Navy at the site (Habitat Restoration Group [HRG], 1993a; TTFW, 2004), and areas were designated as jurisdictional wetland based on evaluation of soil types, hydrologic characteristics, and vegetation types. The earliest wetland delineation concluded that 16.9 acres of wetland existed along with approximately 5.4 acres of open water (HRG, 1993a). The most recent wetland delineation study (TTFW, 2004) further defined on-site wetlands as either salt marsh or seasonal wetland. Figure 2-2 shows areas that have been identified as such at IR Site 2.

Salt marshes are defined as coastal marshes, which are dominated by a prevalence of pickleweed (*Salicornia rubra*). Distinct salt marsh features are functional impoundments that receive tidal influence, and other salt marsh features are impoundments that do not receive direct tidal influence. Seven salt marsh wetlands (SM1-SM7) were identified at IR Site 2 (TTFW, 2004), one of approximately 1 acre and located in the northeastern portion of the landfill and six others consisting of approximately 23 acres and located in the wetland area of the site (see Figure 2-2). Seasonal wetlands are defined as those features that are dominated by a prevalence of certain grasses and grass-like plants (e.g., salt grass [*Distichlis spicata*] and rabbit's foot grass [*Polypogon monspeliensis*]), and are seasonal, meaning they are wet periodically throughout the year. Seven seasonal wetland areas (SW4-SW9 and SW12) were identified at IR Site 2 (Figure 2-2), three consisting of approximately 3.35 acres and located in the northeastern portion of the site and four consisting of approximately 0.25 acre and located in the wetland area of the site. It has not been determined whether these seasonal wetlands are within the jurisdiction of the United States Army Corps of Engineers (USACE).

In this RI Report, the term "wetland" is used to describe those portions of the site characterized as periodically saturated, but not to a degree that results in significant long-term inundation or surface ponding (i.e., open water habitat). Open water areas (i.e., the North and South Ponds) that exist within the overall bounds of the wetland portion of the site and which also meet the functional definition of wetlands, are described as "wetland ponds". Periodic inundation of the wetland areas can be attributed to three primary mechanisms: seasonal ponding of precipitation; water from San Francisco Bay entering the wetlands through the pipe culvert that penetrates the perimeter berm and seawall; and groundwater discharge to the surface (PRC, 1994). Because of variable surface topography within the designated wetlands, some areas

may be less saturated and contain vegetation similar to upland terrestrial vegetation. The distribution of wetland habitat at the site has been described and documented several times in the past (HRG, 1993a; PRC, 1994; PRC, 1996b; FWEC, 2003), and this historical information suggests that the distribution of wetland habitat may vary seasonally and from year to year. Historically, a variety of plants, invertebrates, birds and mammals have been observed in the wetland areas of the site.

A benthic survey was performed at four stations within the wetlands in 1993. In addition, invertebrates were collected at two wetland locations in 1998. Invertebrates found in the wetlands include annelid worms such as oligochaetes and *Polydura cornuta*, crustaceans such as *Allorchestes augusta*, *Ampelisca abdita*, and *Hemileucon hinumensis*, and mollusks such as *Gemma gemma* and *Tellina modesta* (PRC, 1994, as cited in PRC, 1996b; TtEMI, 1998 unpublished data).

A bird survey conducted in 1994 (PRC, 1994) indicated that breeding bird diversity at the site is low due in part to significant flooding during the breeding season and relatively high salinity (HRG, 1993a). However, migrating and wintering diversity is considered high because the site is within the Pacific Flyway and provides over 5 acres of habitat (HRG, 1993a). Approximately 25 shorebird and waterfowl species historically have been observed at the site (PRC, 1994).

Species observed foraging or breeding in the wetlands include the mallard (*Anas platyrhynchos*), black-necked stilt (*Himantopus mexicanus*), American avocet (*Recurvirostra americana*), Caspian tern (*Sterna caspia*), and killdeer (*Charadrius semipamatus*) (PRC, 1994). In addition, special-status species such as the Canada goose (*Branta Canadensis*), the long-billed curlew (*Numenius americanus*), and the Alameda song sparrow (*Melospiza melodia pusillula*) also have been observed using the wetland areas at the site. Mammals that may live, forage and/or reproduce in the wetland habitat include the red fox, black-tailed hare, raccoon, and striped skunk. Although the salt marsh harvest mouse might be expected at the site based on the observed habitat, surveys conducted specifically to detect its presence found no indication of this species at the site. Moreover, due to the limited amount of true salt marsh at the site, the isolated nature of the wetland, and the likely future land use, it is unlikely that this species will be present in the future.

2.10.3 Wetland Ponds

The open water resources in the wetlands consist of two perennial surface water ponds known as North Pond and South Pond (see Figure 2-2). North Pond is connected to the bay by a 36-in-diameter pipe culvert that penetrates the perimeter berm and the seawall. The South Pond is a constructed feature created through the removal of soil material that was subsequently used for landfill cover at IR Site 2. The South Pond is not generally hydrologically connected to the North Pond or to San Francisco Bay. Although not likely based on the geomorphology of the wetland ponds and the surrounding wetland habitat, during times of significant precipitation (i.e., storm events), it is possible that surface water levels could rise to the extent that the wetland ponds may become interconnected or the South Pond may become connected to the bay. The wetland ponds have been observed to retain some amount of water throughout the year, although they have historically been observed to go dry in some years. The South Pond is historically more prone to drying due to its lack of connection to San Francisco Bay. As with the wetlands, the wetland ponds are likely recharged from a combination of three sources: ponding of precipitation; tidal influx from San Francisco Bay through the pipe culvert; and groundwater discharge to the surface (PRC, 1994).

The salinity of the wetland ponds likely varies from being brackish to hypersaline. The northern wetland pond is hydraulically connected to San Francisco Bay, whereas the southern wetland pond is primarily influenced by precipitation and evaporation. As such, the salinity in the wetland ponds is expected to be variable (i.e., salinity in the northern wetland pond is anticipated to be controlled primarily by tidal flush-

ing and precipitation/evaporation patterns, whereas salinity in the southern wetland pond is anticipated to be controlled primarily by precipitation/evaporation patterns). Historical and more recent conductivity and salinity data for the two wetland ponds indicates that the South Pond has similar or higher salinity than North Pond during some portions of the year, which could be related to limited flushing of salts from this more confined feature.

Although vegetation is not typically present in the open water portions of the wetland ponds, the borders of the wetland ponds support cordgrass, and other wetland plants similar to those described above. Existing data indicate that the diversity and abundance of the benthic and aquatic invertebrate communities in the ponds are relatively low. Historical benthic community surveys have revealed that the primary species observed in the wetland ponds are oligochaetes. Other species observed in small numbers include ostracods, amphipods, tenipeds, copepods and clams. The low invertebrate diversity and abundance could be related to predation, seasonal drought conditions, a natural condition, or some combination of such factors. As described above, amphibians and reptiles have not been surveyed at the site, but are expected to occur mainly in the upland areas. Fish historically observed in the wetland ponds are limited to only two species, namely topsmelt and threespine stickleback. Small fish representing a small number of fish species have been collected during historical sampling activities. More recent collection efforts during the RI failed to generate any benthic/aquatic invertebrates or fish samples from the ponds.

It is presumed the birds that may use the wetland ponds are the same as, or a subset of, the birds observed in the wetlands area during the 1994 and 1997 bird surveys. A number of avian species have been observed specifically within "open water" habitat at the site, including the lesser scaup (*Aythya affinis*) and the surf scoter (*Melanitta perspicillata*). The complete list of species historically observed can be found in Table 2-3. All birds discussed herein are assumed, for the purpose of this RI, to use both wetland ponds. No mammals have been identified at the site that would preferentially use open water wetland pond habitat for foraging. However, the raccoon is an opportunistic omnivore and may eat aquatic species such as insects, frogs, and crayfish. The raccoon also may forage general wetland habitats and will consume virtually any plant or animal matter (U.S. EPA, 1993).

2.11 Potential Sources of Contamination

Wastes were disposed across the entire area of the IR Site 2 landfill during its period of operation. The types of wastes disposed at the landfill were numerous, and, in general, there was only minimal segregation of the wastes before they were deposited into the landfill. The general waste placed in the landfill consisted of many typical municipal wastes, including household garbage, food waste, and paper products. To aid in identifying potential contamination source areas (i.e., hot spots), historical information from the IAS (E&E, 1983) was used to develop a map identifying discrete wastes that were disposed at IR Site 2. The locations of these discrete waste disposal areas were mapped during the IAS, which in turn was based largely on a review of available facility records and interviews with facility personnel. It is recognized that inherent uncertainties are associated with historical facility records and personnel interviews. However, the information provided herein is true and accurate to the extent that the historical information relied on is true and accurate. The discrete disposal areas are shown on Figure 2-11 and summarized briefly in following paragraphs. The physical boundaries associated with the discrete disposal areas shown on Figure 2-11 are meant to convey the general locations of these discrete waste disposal areas, and not to suggest that they are completely delineated in extent.

- **Chemical Drum Disposal Areas:** Disposal of chemical drums took place in four areas reportedly termed the "chemical dump". Specifically, these four areas were located at the northeastern corner, mid-eastern side, mid-northern area, and mid-western side of the landfill. Figure 2-11 outlines the approximate areas of chemical drum disposal based on the historical information from the IAS (E&E, 1983). Chemical drums were reportedly

disposed of in the landfill by first excavating a trench down to the water table, placing full chemical drums into the trench, puncturing and draining the drums, and filling the trench with soil (E&E, 1983). For some period of time, drums were reportedly disposed whole (i.e., without puncturing), but after the occurrence of three landfill fires related to this practice, the general practice was altered to include puncturing and draining during disposal. The quantity of drums and associated liquid wastes disposed is unknown and there is no record that the drums were ever removed from the site.

- **Pesticide Disposal Area:** In the late 1970s, the Public Works Pest Control Shop (i.e., IR Site 8) removed its inventory of pesticides subject to the Toxic Substances Control Act (TSCA) and the Federal Insecticide, Fungicide, and Rodenticide Act (FIFRA), and disposed of these pesticides in the landfill (see Figure 2-11). The pesticides (reportedly both solids and liquids) were contained in cardboard, glass, and plastic containers, and were reportedly disposed of in the southeastern portion of the landfill. The pesticide disposal area was approximately 30 to 50 ft on a side (i.e., 900 to 2,500 ft^2). The pesticides were reportedly disposed of in a ditch which was covered with 3 to 4 ft of soil (E&E, 1983). The volume of pesticides disposed of in this location is unknown and no records indicate that the pesticides were ever removed from the site.

- **Asbestos Disposal Areas:** Asbestos pipe lagging historically removed from ships was disposed of in the mid-eastern section of the landfill (see Figure 2-11). The asbestos was transported loose to the landfill in dumpsters until the early 1970s, after which it was put into plastic bags prior to disposal (E&E, 1983). The volume of asbestos waste deposited is unknown and there are no records indicating that the waste was ever removed from the site.

- **OEW Burial Site:** An approximately 2.5-acre area known as the Possible Ordnance and Explosives Waste (OEW) Burial Site is located in the southeast corner of the landfill (see Figure 2-11). This area was first identified by Explosive Ordnance Disposal (EOD) personnel from Supervisor of Shipbuilding, Conversion and Repair, Portsmouth (SSPORTS) as a possible ordnance burial site. The identification of this area was based on the results of a geophysical survey, site history, and interviews conducted with former NAS Alameda departmental personnel. Reportedly, the OEW originated from the Defense Logistics Agency in Alameda. Approximately four truckloads of OEW ranging in size (from 4-ft-long by 1-ft-wide to smaller munitions) were disposed at this location in 1976 (SSPORTS, 1999a). In 2002, a total of 8,675 20-mm soft steel target practice rounds were identified within 12 in of the surface at the burial site, and were removed in accordance with an emergency removal action (FWEC, 2003).

- **Unlined Oil Pits:** Two unlined pits were reportedly located at the landfill for waste oils that were not reclaimed or sold (see Figure 2-11). It was reportedly a standard practice for oil tankers to drain oil into the pits. After the oil tankers were finished, there was typically a visible sheen on impounded water located in the midsection of the landfill. The pits were reportedly located in the northeastern corner of the landfill and in the southwestern corner, just south of the approximate boundary between the landfill and the wetland along the western perimeter of the site (E&E, 1983). The volume of oil disposed of in the pits is unknown and no records indicate that the oil was ever removed from the site.

- **Radioactive Waste Storage Shack:** A former radioactive waste storage shack located in the northern portion of the landfill was investigated by SSPORTS in 1999 (see Figure 2-11). During a radiological survey, eight anomalies were identified with a maximum activity of 2.7 microcuries (μCi) (SSPORTS, 1999b). Surface soil around the shack was shown to contain radium[226] (Ra-226) at levels greater than study-specific reference/background levels. All potential radioactive sources that were identified within 12 in of the surface were removed (SSPORTS, 1999c).

- **Dredge Spoil Spreading Area:** The southern portion of the site served as a location for the deposition of dredge spoils from Oakland Inner Harbor, the pier area and turning basin of Alameda Point, and Seaplane Lagoon (see Figure 2-11). These dredge spoils were in turn used for landfill cover during landfill closure operations. The wetland was primarily used for stockpiling dredged material from the pier areas, turning basin and the harbor entrance channel. In 1981, 24,000 yd^3 of dredged material from the southern side of the Seaplane Lagoon were reportedly deposited in this area (E&E, 1983). Based on recent investigation activities at Seaplane Lagoon, sediments from this location could have contained elevated levels of organic (i.e., PCBs and pesticides), inorganic (i.e., metals), and radioactive (i.e., radium) contaminants. However, the available data from Seaplane Lagoon suggest that contamination at that site is limited to the northern portion of the lagoon, whereas the dredged material deposited at IR Site 2 reportedly originated from the southern portion of the lagoon. Based on characterization activities completed at the Oakland Inner Harbor and the pier area and turning basin, contaminants potentially originating from those locations would likely have been metals, PCBs, polycyclic aromatic hydrocarbons (PAHs), and/or pesticides.

- **Scrap Metal Disposal Area:** Historical documentation indicates that scrap metal from Public Works Center (PWC) sites was disposed of in the wetland area in the late 1970s (E&E, 1983). The quantity of scrap metal disposed of at the site is unknown and there are no records indicating that the scrap metal was ever removed from the site.

- **PCB-Containing Oil:** Oil from transformers that potentially contained PCBs was reportedly spread on the roadways in the northeastern corner of the landfill (see Figure 2-11). In addition, PCB-contaminated rags and PCB-containing carbonless paper were reportedly disposed of in the landfill. Reportedly, approximately 1,400 tac rags (3-ft by 3-ft) soaked with PCBs were disposed of each week, beginning in the early 1950s and continuing through the 1970s. The tac rags also were potentially contaminated with radium from the instrument dial painting and stripping shop. Tons of PCB-containing carbonless paper used on base also were disposed of at the landfill (E&E, 1983). The volume of oil spread on the roads is unknown, and no records indicate that the tac rags, carbonless paper, or oil were ever removed from the site.

- **Landfill Roadways:** Roads on the landfill were reportedly constructed in part from spent sandblasting grit and abrasives from refinishing of metal parts and blasting of ship hulls. It also reportedly was a common practice to spread oily material onto the roads at the site for dust suppression by directly discharging from moving tanker trucks.

2.12 Proposed Future Land Use

In accordance with conceptual plans prepared by the Alameda Reuse and Redevelopment Authority (ARRA), which is the local reuse authority for all property conveyance, planning, and reuse implementation at Alameda Point, the anticipated future use for IR Site 2 is recreational.

3.0: PREVIOUS ENVIRONMENTAL INVESTIGATIONS

This section provides a summary of previous investigations performed at IR Site 2 from 1990 to present. The subsections summarize the previous investigations for each affected media type including soil, soil-gas, groundwater, surface water, sediment, porewater, and tissue sampling that has been conducted at the site. OEW and geotechnical characterization studies are summarized, as well as reference and background investigations that have been performed.

3.1 Summary of Previous Investigations

Field investigations that have been performed at IR Site 2 include:

- Phases 1 and 2A solid waste assessment test (SWAT) activities conducted in 1990;

- Phases 5 and 6 SWAT activities conducted in 1991;

- Ecological assessment conducted in 1993;

- Wetland evaluation technique (WET) analysis conducted in 1993;

- Additional field activities conducted in support of the ecological assessment in 1994 and 1995;

- Radiological surveys conducted from 1995 to 1999;

- Supplemental ecological investigation conducted in 1996 and 1997;

- Continuous and on-going groundwater monitoring starting in 1991;

- Biological sampling conducted to support the ecological risk assessment in 1998;

- OEW survey and removal action conducted in 2002 and 2003; and

- Geotechnical and seismic evaluations conducted in 2002 and 2003.

A description of the sampling activities and a general description of the data generated for each media/waste are provided below.

3.1.1   General Surveys

Kister, Savio, and Rei, Inc. (KSR), licensed land surveyors in the State of California, performed site surveys at IR Site 2 to establish control during geotechnical and seismic evaluations performed by FWEC (2003). The survey control for the site was based on a monument located at the northwest corner of Main Street and Atlantic Avenue in the City of Alameda. The site coordinates are based on the California Coordinate System (CCS) Zone III, North American Datum (NAD) of 1927. The site elevations are based on National Geodetic Vertical Datum (NGVD) of 1929.

A topographic survey also was performed as part of geotechnical and seismic evaluations performed by FWEC (2003). HJW Geospatial, Inc. constructed a topographic map of IR Site 2 using computer-assisted, photogrammetric methods and aerial photographs taken in March 2002. Topographic contours were based on CCS Zone III, NAD27 and elevations were based on NGVD29. The topography of the site that was identified through this previous survey is shown in Figure 2-2.

In January 2002, a USACE Class I hydrographic (bathymetric) survey was performed by EcoSystems Management Associates, Inc. The survey was performed in areas that were accessible by survey vessel and extended approximately 500 ft offshore. Survey lines were established normal to the general

shoreline orientation at 50-ft intervals. Tie lines were set up to intersect the survey lines at approximate 100-ft spacing from the shoreline to the offshore limit of the survey area.

3.1.2 Waste Characterization Investigations

Previous investigations have been conducted at IR Site 2 to characterize the type and extent of waste that was disposed while landfilling occurred. These waste characterization investigations were in the form of geophysics to locate ferrous objects (e.g., buried drums and potential OEW), test pitting to inspect the integrity of buried waste and to determine the thickness of landfill cover, OEW screening, and radiological investigations. The following subsections provide a summary of the previous waste characterization investigations that have been performed at IR Site 2.

3.1.2.1 Geophysical Investigations

During the Phase 1 and 2A SWAT activities (1990), an attempt was made to classify stratigraphy in the subsurface by performing a transient electromagnetic (TEM) survey. Soundings were performed along two north/south and two east/west transects across IR Site 2 at 200-ft intervals. The results were analyzed to determine the vertical and horizontal distribution of subsurface electrical properties. A magnetic survey also was performed at the West Beach Landfill during the Phase 1 and 2A SWAT activities (1990), in order to locate ferrous objects that might be buried at shallow depths. IR Site 2 was divided into two separate areas, one designated as the northern area, which covered approximately 40 acres, and the other referred to as the southern area, which covered approximately 4 acres. Measurement stations were located on a rectangular grid spacing of 20 ft. The northern area contained 4,200 stations and the southern area contained 444 stations. Data collected from the study were plotted on a site map to locate magnetic anomalies. More details regarding this investigation are documented in the *Solid Waste Water Quality Assessment Test Data Summary Report, Remedial Investigation/Feasibility Study Phase 1 and 2A* (PRC and Montgomery Watson, 1993).

An approximate 2.5-acre area located in the southeastern portion of IR Site 2 was identified by SSPORTS OEW personnel as a possible OEW burial site. The identification of the site was based on the geophysical survey conducted during Phase 1 and 2A SWAT activities (1990), as well as interviews with the Alameda Point Weapons Department. A geophysical survey was performed by SSPORTS OEW personnel in 1998 in an attempt to specifically identify large subsurface masses and anomalies that could potentially be ordnance. However, due to the large amount of subsurface debris present and high background noise, attempts were not successful. Detailed results from the survey are located in the *Unexploded Ordnance Site Investigation Final Summary Report* (SSPORTS, 1999a).

3.1.2.2 Test Pits

Twelve (12) test pits were excavated during OEW/geotechnical characterization activities in 2002 (FWEC, 2003) to determine the cover thickness on the landfill. These test pit locations are shown in Figure 3-1. OEW technicians cleared each test pit location of metal debris by scanning the area with a Schonstedt magnetometer. After an area clear of metal debris was located, the soil was mechanically removed in 1-foot lifts. The OEW technicians checked each lift with the Schonstedt magnetometer, and if metal debris was located, it was excavated by hand and confirmed to not be ordnance-related. Each test pit was excavated to a maximum depth of 4 ft bgs. Bulk geotechnical soil samples were collected from the excavated soil. The excavated soil also was classified in the field using American Society for Testing and Materials (ASTM) Test Method D2488. Moisture content, water flow, waste encountered, if any, and significant odors also were noted during the investigation.

Results of the test pit explorations are summarized in Table 3-1. These explorations revealed that soil cover thickness over the landfill varied from approximately 2 inches to 2 ft. In general, no liner was observed, and less than 2 ft of soil cover consisting of fine-grained materials existed in most areas. Refuse included materials such as concrete, asphalt, brick, pipe, wood, plastic, hoses, and metal objects, and was observed in all of the test pits. No significant odor was detected in any of the test pits, but some soil discoloration was observed in TP-2-1, TP-2-6, TP-2-7, TP-2-8, TP-2-9, and TP-2-11. Most soil samples collected from the test pits were moist likely from rainy weather conditions and close proximity to the bay. Significant water seepage was observed at 2.5 to 3.0 ft bgs when excavating TP-2-10 and TP-2-12. The existing soils cover was found to be inconsistent, poorly compacted, and very permeable; therefore, it is likely unsuitable to be considered part of a final cover design.

3.1.2.3 Ordinance and Explosives Waste

An OEW surface screen was performed in the landfill area during OEW/geotechnical characterization activities performed in 2002 (FWEC, 2003). The IR Site 2 landfill was divided into 200-ft by 200-ft grid sections that were swept individually. The OEW team cleared each grid by sweeping a Schonstedt GA-52 CX in small arcs in front of them. When OEW was encountered, the location was assigned a northing and easting identification that was measured from the southwest corner of the individual grid so it could be plotted on a site map using the Geographic Information System (GIS) of Alameda Point. One M56 anti-tank/anti-personnel (AT/AP) inert land mine and one 20-mm target practice projectile were found during the surface characterization of IR Site 2.

Following the OEW surface screen, a 20-ft by 20-ft grid was established in the possible OEW burial site (see Figure 3-1) to identify the location of subsurface OEW and perform a Time Critical Removal Action (TCRA) (FWEC, 2003). A total of 8,675 20-mm target practice projectiles were uncovered. None of the OEW encountered at IR Site 2 contained any explosives or energetics. All 20-mm target practice projectiles were demilitarized by cutting them in half prior to disposal of as inert metal in a Class III Landfill. The AT/AP inert land mine that was discovered during the surface screen was returned to the Navy OEW Detachment Southwest Unit at Building 41, NAS North Island. A complete discussion of the TCRA can be found in the *Final Time-Critical Removal Action Closeout Report* (FWEC, 2002).

3.1.2.4 Radiological Investigations

The IAS (E&E, 1983) reports that the use of radioactive materials at the site began in 1940, particularly radium at the dial-painting shop section of the instrument shop at Building 5. Radium-impacted wastes including scraping solids, rags, and used paint brushes from refurbishing dials and gauges reportedly were collected from the shop on a regular basis and discarded at IR Sites 1 and 2. The radium paint shop was closed in the early 1960s and apparently work materials that could not be decontaminated were placed in the IR Site 1 landfill. Recall disposal began to occur at IR Site 2 around 1956, suggesting there was approximately 5 years of potential radium waste disposal that occurred at IR Site 2. The potential for radium waste being present at Site 2 has resulted in the completion of several radiological investigations as summarized in Table 3-2.

The radionuclide of concern from historic dial-painting operations is Ra-226. As such, radiological investigations completed historically at IR Site 2 have focused on this compound only. PRC performed two radiological investigations, in 1995 and 1996. The initial September 1995 investigation was performed in order to identify near-surface radiological contamination in the landfill. The 1996 investigation included additional surveys that were performed from May to September and covered the areas of the coastal margin, interior margin, and landfill. Between the 1995 and 1996 investigations, 40 radiological anomalies were identified. The results of this investigation are summarized in *Radiation Survey Report, Naval Air Station Alameda, Alameda, California* (PRC, 1997a).

A more comprehensive survey was performed by SSPORTS (1999b) Environmental Detachment in 1998 and 1999. A scanning survey was conducted using a four detector array of 3"× 3" shielded sodium iodide detectors in flat areas and a single 2"× 2" detector in areas that were difficult to access. The surveys covered 100% of the 15.7 acres included in the study. A total of 951 points were identified as having radiation of at least twice the normal background level. Radiological removal actions were conducted at 50 points with radiation counts greater than 4 times the study-specific local background (SSPORTS, 1999c).

3.1.3   Surface Water Investigations

Surface water investigations were conducted at IR Site 2 in 1991, 1993, and 1996-97 to determine whether surface waters in the wetland ponds were being impacted by the landfill via surface water runoff, direct leaching from the landfill, groundwater transport, or airborne dispersion and deposition of chemically affected particles (PRC and Montgomery Watson, 1993; PRC and TtEMI, 1992a and 1992b; PRC 1996c). Table 3-3 summarizes each of the historical surface water sampling investigations and lists the parameters that were analyzed for the surface water samples. Figure 3-2 shows all of the historical surface water sampling locations.

During SWAT Phases 5 and 6 performed by PRC and Montgomery Watson in May 1991, 50 surface water samples were collected from 23 locations in the North and South Ponds. PRC collected an additional 5 surface water samples from 7 locations during a follow-up investigation in 1996 and 1997. In 1998, TtEMI performed additional surface water investigations to determine the potential for transport of soil-bound chemicals suspended in storm water runoff from the landfill into the ponds. Data collection included water level and water quality measurements designed to determine the effects of storm events and the San Francisco Bay tidal cycle on wetland ponds. A total of 30 surface water samples were collected from 3 locations (SP-E in the South Pond and NP-E and NP-C in the North Pond) in February and March 1998. A total of 85 surface water samples have been collected from the North and South Ponds during historical investigations.

3.1.4   Sediment Investigations

Sediment samples were collected at IR Site 2 during three separate events in 1991, 1993-1994, and 1996-1997 to determine if soil from the landfill was being transported to the ponds by either surface water runoff or airborne dispersion, or if the original dredge material that was used to build the wetland area had been impacted by chemicals. Table 3-4 summarizes each of the historical sediment sampling investigations and lists the parameters that were analyzed for the sediment samples. Figure 3-3 shows all of the historical sediment sampling locations.

During SWAT Phases 5 and 6 (1991), a total of 13 sediment samples were collected from the wetland ponds. PRC collected an additional 20 sediment samples from the wetland ponds in 1993. In 1996 and 1997, sediment samples also were collected by PRC during a follow-up ecological assessment investigation. In all, 39 sediment samples have been collected from the wetland ponds at IR Site 2.

3.1.5   Porewater Investigations

Pore water sampling has been done in the past to determine whether sediment bound chemicals are potentially desorbing from sediment and diffusing into the surface water of the wetland ponds. Table 3-5 summarizes the historical porewater sampling investigations and the parameters that were analyzed for the porewater samples. In 1996, 41 porewater samples were collected from three locations in the North Pond, and in 1997, 48 porewater samples were collected from three locations in the South Pond.

3.1.6 Geotechnical Investigations

Geotechnical investigations were performed during SWAT Phases 1 and 2A, Phases 5 and 6, field activities in 1994-1995, and for 2002 OEW/geotechnical characterization work. In May 1990, 21 samples were collected from 4 monitoring well installation borings for geotechnical analyses including moisture content, dry density, wet density, specific gravity, permeability, grain size distribution, consolidation, liquid limit, plastic limit, plasticity index, and compaction. Twenty-six additional geotechnical samples were collected from 23 monitoring well borings during SWAT Phases 5 and 6. These samples were submitted for moisture content, dry density, specific gravity, cation exchange capacity, permeability, grain size distribution, and consolidation. Additional geotechnical investigations in 1994 included collection of 4 more samples that were analyzed for moisture content, dry density, specific gravity, cation exchange capacity, permeability, grain size distribution, liquid limit, plastic limit, and plasticity index.

In February and March 2002, a geotechnical evaluation was performed at IR Site 2 to identify potential hazards for the Feasibility Study (FWEC, 2003) and determine the thickness of the existing soil cover over the landfill. The field investigation included 21 CPT borings advanced using a 20-ton CPT rig, 12 test pits excavated using a backhoe, and 15 soil borings (9 upland and 6 offshore) installed using a mud rotary system. Representative disturbed and relatively undisturbed soils samples were collected from various depths in surface and subsurface soils, and analyzed for geotechnical-related tests as summarized in Table 3-6. Immediate and long-term settlements resulting from a landfill cap assumed to be installed at the site in the future were estimated using the theory of elasticity and one-dimensional consolidation theory (Terzaghi as described by Coduto, 1994). The maximum ground settlement expected to occur from an assumed 4-foot landfill cap is approximately 13 inches. These settlements are expected to occur over a long period of time (e.g., 40 years or more), thus they do not pose an immediate hazard. The static stabilities of various slope cross-sections were analyzed to obtain factors of safety against slope failure. This factor is defined as the ratio of resisting (stabilizing) forces to the driving forces trying to displace the slope. Guidelines for the stability analyses are provided in Title 27 CCR; however, no specific value for the static factor of safety is provided. The current state of practice in California at the time the seismic hazards analysis was performed was to require a minimum static factor of safety of 1.5. Six different cross sections across IR Site 2 were analyzed with an assumed 4-foot-thick soil cover, and all except one were determined to have a static factor of safety greater than 1.5. The factor of safety for the lone cross section that was below 1.5 was 1.46, which led to a conclusion that some remedial measures involving geotechnical improvements might be needed to increase the static factors of safety to meet the California standard of practice. The thickness of the existing soil cover was found to vary from approximately 2 inches to 2 ft over the refuse in the landfill, with less than 2 ft of cover in most areas.

3.1.7 Surface Soil Investigations

Surface soil sampling activities have been performed at IR Site 2 during three separate investigations to characterize potential contamination. Table 3-7 summarizes each of the historical surface soil sampling investigations and lists the parameters that were analyzed for the samples. Note that some geotechnical data were collected from surface soils during OEW/geotechnical characterization work described in Section 3.1.5 (FWEC, 2003) and are not described here. Figure 3-4 shows all of the historical surface soil sampling locations.

During SWAT Phases 1 and 2A (1990), a total of 10 surface samples (i.e., 0 to 2 ft bgs) were collected from four soil boring locations (WB-1 through WB-4). During SWAT Phases 5 and 6, IR Site 2 was divided into grids of 200 ft^2, and 168 surface soil samples were collected from 151 locations within the grid (1 location in each square plus 17 duplicate samples). The rationale for this systematic surface soil sampling was to determine if chemical contamination from the landfill may have translocated to the

surface cover of the landfill, and whether the landfill cover contained higher concentrations of chemicals than soils in other areas of IR Site 2 (e.g., wetland, and interior and coastal margins).

Surface soil samples also were collected from soil borings in the landfill and the coastal margin in conjunction with the installation of groundwater monitoring wells. Three surface soil samples were collected during monitoring well installation in 1990, 13 samples were collected in 1991, three samples were collected in 1994, and 8 samples were collected in 1995 (Canonie, 1990a and 1990b; PRC and Montgomery Watson, 1993). Sixteen additional surface soil samples were collected by PRC and Montgomery Watson during the installation of 15 monitoring wells in November/December 1990 and May 1991.

In 1995, eight surface soil samples were collected from seven borings locations to support an *Ecological Assessment Work Plan* (PRC and Montgomery Watson, 1993). Two boring locations were located in the landfill, three were in the coastal margin, one was in the wetland area between the North and South Ponds, and one was in the interior margin. These eight samples were analyzed for total petroleum hydrocarbons (TPH) only.

3.1.8 Subsurface Soil Investigations

Subsurface soil investigations were performed during SWAT Phases 1 and 2A (1990), SWAT Phases 5 and 6 (1991), and the 1994-1995 ERA to characterize lithologic and geotechnical properties of the soil, and to determine the vertical and horizontal extent of chemical concentrations in soils. Note that some geotechnical data were collected from subsurface soils during the OEW/geotechnical characterization work described in Section 3.1.5 (FWEC, 2003) and are not described here. Table 3-8 summarizes each of the historical subsurface soil sampling investigations and lists the parameters that were analyzed for the samples. Figure 3-5 shows all of the historical subsurface soil sampling locations.

During SWAT Phase 1 and 2A, 36 subsurface soil samples were collected from four soil borings for chemical analyses, and an additional 21 samples were collected from these borings for geotechnical analyses. During SWAT Phases 5 and 6, 23 subsurface soil samples were collected from 18 monitoring well borings for chemical analyses, and an additional 26 samples were collected for geotechnical analyses. In August/September 1994, six subsurface soil samples were collected from three monitoring well borings for chemical analyses, and an additional four samples were collected for geotechnical analyses. In 1995, 20 subsurface soil samples were collected from 12 monitoring well borings for chemical analysis.

3.1.9 Groundwater Investigations

Groundwater characterization began at IR Site 2 in 1991 to determine if any chemical impacts occurred in the landfill, and were potentially migrating offsite (Canonie, 1990a and 1990b; PRC and Montgomery Watson, 1993). A total of 42 monitoring wells have been installed during historical investigation activities: 27 in 1991, 3 in 1994, and 12 in 1995; 29 wells are screening in the FWBZ and 13 are screened in the SWBZ. Figure 3-6 shows the locations of permanent groundwater monitoring wells at that site.

In 1991 and 1992, 132 groundwater samples were collected from 29 monitoring wells during three separate sampling events consisting of June 1991 to August 1991, September 1991 to October 1991, and January 1992 to April 1992. During quarterly monitoring in 1994 (October) and 1995 (February, May, and July), 100 groundwater samples were collected from 24 permanent monitoring wells, in addition to 16 Hydropunch™ groundwater samples from 12 locations. Five groundwater monitoring events occurred from 1996 to 1998, in January 1996, February 1997, October/November 1997, February 1998, and May 1998. Various analyses were performed during these historical groundwater sampling events and consisted of metals, PAHs, SVOCs, VOCs, PCBs, pesticides, TPH, radionuclides, total organic carbon

(TOC), cyanide, sulfide, solids, acidity, hardness, chemical oxygen demand (COD), pH, alkalinity, anions, and conductance.

Currently 42 monitoring wells are included in the Navy's quarterly groundwater monitoring program that has been in place at IR Site 2 since summer 2002. A portion of the monitoring wells are sampled on a quarterly basis whereas others are sampled semiannually. The monitoring program is reviewed and optimized on an annual basis to ensure that resources are focused on continuing to meet data quality objectives (DQOs) at the site. Table 3-9 summarizes the analyses that are currently performed by ITSI (2005) during quarterly groundwater monitoring events.

3.1.10 Soil-Gas Investigations

Soil-gas sampling began at IR Site 2 as part of the groundwater monitoring program that began in 2002. Five soil-gas monitoring points with screens at 2 or 3 different depths each were installed within the landfill in September 2002. Figure 3-6 shows the locations of the soil-gas monitoring probes, and indicates the number of screens at each location in parenthesis (S = shallow, M = medium, and D = deep). Table 3-10 summarizes the soil-gas monitoring parameters that are measured and the frequency at which the probes are sampled.

3.1.11 Ecological Investigations

Various types of ecological investigations have been conducted at IR Site 2 including wetlands characterization activities, various biological surveys, and tissue sampling. The following subsections summarize these historical ecological investigations.

3.1.11.1 Wetlands Characterization

The first wetlands delineation activities that were performed at IR Site 2 occurred in February and March 1993. The initial studies delineated jurisdictional wetlands and conducted a WET analysis within the wetlands, the results of which are described in detail in the draft ecological assessment report (PRC, 1994). These initial analyses indicate that the amount of open water within the North and South Ponds varies with season and rainfall, and the soil, which occurs in depressions of the wetland or in areas that are ponded for a long duration during the growing season, can be classified as hydric. It was inferred that the water in the wetland area seemed to originate from three sources: seasonal ponding of precipitation; tidal Bay water entering the area through a culvert on the western boundary of the North Pond; and potential gains from groundwater. The salinity of the pond water in the wetland was found to be generally greater than 30 parts per thousand (ppt) (HRG, 1993a).

In November 2001, FWEC biologists conducted a wetland assessment to determine the potential impacts on wetland and water resources during the OEW/geotechnical characterization activities (FWEC, 2003). This biological study was performed to identify the location and boundaries of all jurisdictional wetland and waters within the proposed work area subject to jurisdiction by the USACE under Section 404 (b)(1) of the Clean Water Act. The results of the study indicated the presence of three wetland areas, one consisting of approximately 29.3 acres of salt marsh wetland habitat in the southwestern portion of the site, and the other two consisting of approximately 0.23 acre of seasonal wetland habitat in the northeastern portion of the site. Hydric soils were determined to exist in each of these wetland areas. In addition to the biological studies, avian inspections were performed prior to the planned field activities to minimize adverse impact to bird species listed as Endangered, Threatened, or Candidate species under federal and California state laws, as well as to certain other species which receive protection under the California Department of Fish and Game (CDFG) codes. The wetland studies and avian inspections found that the planned field activities did not result in the permanent loss of any jurisdictional wetlands and no active

nests protected under federal and California state laws and CDFG Code Sections 3503 or 3503.5 were identified. Refer to the *Draft Final OEW/Geotechnical Characterization Report* (FWEC, 2003) for additional details associated with these studies.

The most recent wetlands delineation activities were performed by the Navy (TTFW, 2004) as part of the radiological surveying activities at IR Sites 1 and 2 in 2004. In September 2004, the Navy conducted a detailed wetland delineation to determine the potential extent of Waters of the United States per Section 404 of the Clean Water Act, as regulated by the USACE. All of the delineation work was performed in accordance the *Wetlands Delineation Manual* (USACE, 1987), *Clarification and Interpretation of the 1987 Manual* (USACE, 1992), and the January 9, 2001, U.S. Supreme Court-issued decision for the Solid Waste Agency of Northern Cook County. The progression of the delineation activities were as follows: initially vegetation was evaluated, and if it met the USACE established criteria, then soils were examined; if the vegetation and soils were consistent with the USACE established criteria, then hydrology data were evaluated. The results of the wetland delineation work indicated that seven salt marsh wetlands exist at the site, one of approximately 1 acre and located in the northeaster portion of the site (in the interior margin) and six other consisting of approximately 23 acres and located in the wetland area of the site. In addition, seven seasonal wetland areas were identified at the site, three consisting of approximately 3.35 acres and located in the northeastern portion of the site (in the interior margin) and four consisting of approximately 0.25 acre and located in the wetland area of the site. For additional details regarding the findings of this wetland delineation study and a map showing their location, refer to Section 2.10.2 and Figure 2-2, respectively.

3.1.11.2 Biological Surveys

Between 1993 and 1997, biological surveys for threatened and endangered species, plants, benthic-invertebrates and avian receptors, and toxicity and bioaccumulation testing were performed at IR Site 2.

3.1.11.2.1 Threatened and Endangered Species Surveys

A threatened and endangered species survey was conducted between 1995 and 1997 by TtEMI to determine the occurrence, or potential for occurrence, of threatened and endangered terrestrial and aquatic species at or near Alameda Point. The survey included both literature reviews and field surveys to identify threatened and endangered species of plants and animals potentially present at Alameda Point. Surveys were conducted for plants, mammals, amphibians, reptiles, and birds. Field trapping surveys were conducted specifically for the endangered salt-marsh harvest mouse.

The results of the literature search and field surveys found that the occurrence of threatened and endangered species is limited to the wetland areas that occur in the southwest quadrant of Alameda Point, specifically at the West Beach Wetland and Runway Wetland. Although the literature search identified several species of plants, invertebrates, fishes, amphibians, reptiles, and mammals that could potentially occur, the industrial nature of Alameda Point and the isolated and disturbed nature of these areas may preclude using these areas as animal habitats. In contrast, a number of threatened and endangered bird species have been observed at Alameda Point, mainly in the wetland areas but also flying over the area or using offshore habitat in the Bay adjacent to the wetland. Threatened and endangered bird species known to occur at Alameda Point include American peregrine falcon (*Falco peregrinus anatum*), western snowy plover (*Charadrius alexandrinus nivosus*), California least tern (*Sterna antillarum browni*), saltmarsh common yellowthroat (*Geothlypis trychas sinuosa*), Alameda song sparrow (*Melospiza melodia pusillula*), and California brown pelican (*Pelicanus occidentalis californicus*). All of these species except the pelican could potentially forage at IR Site 2. Observations of these species were reported by Feeney and Collins (1993) or were made during the 1997 avian surveys conducted for the wetland areas (TtEMI, 1998a) that are summarized in Section 2.10.

3.1.11.2.2 Plant Surveys

Plant surveys were conducted at IR Site 2 as part of the threatened and endangered species field surveys in 1997. The field surveys were performed by systematically traversing the survey area in parallel lines spaced 5 to 10 ft (1.5 to 3 m) apart. Terrestrial plant species were identified and documented to substantiate the presence or absence of any threatened, endangered, or sensitive plant species. The plants that were found at the site, which are summarized in Table 3-11, represent a mix of native and nonnative species with no threatened, endangered, or sensitive plant species being found.

3.1.11.2.3 Benthic-Invertebrate Survey

Benthic-invertebrate community analyses of sediment in the wetland ponds were conducted in 1993 and 1994 to determine whether chemicals present in the wetland were impacting benthic-invertebrate diversity or community structure. Four locations at the wetland were sampled to characterize the benthic-invertebrate community. Sediments were collected and sieved, and invertebrates present in the sediment were identified and catalogued as described in applicable work plans (PRC and TtEMI, 1992a and 1992b).

Typical benthic-invertebrate fauna observed in the wetland ponds include annelid worms such as oligo-chaetes and *Polydura cornuta*; crustaceans, such as *Allorchestes augusta, Ampelisca abdita, Corophium insidiosum, Hemileucon hinumensis* and *Podocopa* sp.; and mollusks, such as *Gemma gemma* and *Tellina modesta* (PRC, 1994). Few species and low densities of individual organisms were found during the benthic-population sampling (PRC, 1994).

3.1.11.2.4 Avian Survey

Between January and May 1997, avian surveys were conducted at the wetland at approximately a bi-monthly schedule to characterize the bird communities present at the site. Ten individual surveys were conducted at the wetland on the following dates: January 13 and 28, February 13, March 3 and 22, April 3 and 18, and May 3, 22, and 29. A detailed description of the survey results are discussed in Section 2.10.

3.1.11.2.5 Toxicity Tests

Solid-phase toxicity tests were conducted on sediments from seven locations in the wetland ponds in 1993 and 1994 (PRC and Tetra Tech Inc., 1992a, 1992b). Five locations were in the North Pond and two were in the South Pond. Survival and reburial of the amphipod *Eohaustorius estuarius* and growth of the polychaete worm *Neanthes arenaceodentata* were measured in five replicate tests conducted on surface sediments from each of the seven sampling locations. The results of the historical sediment bioassays are presented in Table 3-12. *Eohaustorius* mortality exceeded the Bay reference envelope upper threshold limit (UTL) of 67.7% survival at three of the seven bioassay locations (2 in the North Pond and station 1 in the South Pond). *Neanthes* survival was not significantly different from the laboratory control at any of the test stations, and *Neanthes* growth differences were within the Bay reference envelope UTL of 55.4%, indicating that *Neanthes* was not adversely impacted by the wetland sediments.

3.1.11.2.6 Bioaccumulation Tests

Bioaccumulation studies using the clam *Macoma nasuta* and the sea-urchin *Strongylocentrotus purpuratus* were conducted on sediment samples collected from four locations in the wetland ponds (2 in the North Pond and 2 in the South Pond) in 1993 and 1994. These tests were conducted to determine if chemicals sorbed to sediment were bioavailable to benthic organisms and were potentially bioaccumu-lating up through the food chain. The resulting tissue samples were analyzed for metals, PAHs, SVOCs,

PCBs, pesticides, organotins, and moisture content. The only chemicals detected in all *Macoma* tissues exposed to IR Site 2 wetland sediments included arsenic, copper, lead, and zinc. Mercury, Aroclor-1254, 4-methylphenol, and di-*n*-butylphthalate were each detected in 3 of 4 tissue samples, whereas nickel, 4,4-DDE, and dibutyltin were each detected in 2 of 4 *Macoma* samples. Aroclor-1260, 4,4-DDD, *gamma*-chlordane, and *bis*(2-ethylhexyl)phthalate each were detected in one *Macoma* tissue sample. No other constituents were detected in the *Macoma* bioassays.

3.1.11.3 Tissue Investigations

Tissue sampling was conducted in 1996 and 1998 to support an ERA and, specifically, to assist in estimating the potential chemical doses to upper trophic-level receptors. A sampling design was developed to collect plants, invertebrates, and small mammals from the terrestrial habitats of the landfill; and plants, invertebrates, nonmigratory fishes, and small mammals from the aquatic and wetland habitats of the wetland. Additionally, an attempt was made to collect plants, invertebrates, and small mammals from a reference area north of IR Site 2 to evaluate ambient chemical levels in these tissue-sample types. The historical tissue sampling areas for IR Site 2 are shown in Figure 3-7. The following subsections summarize the historical tissues sampling activities that have been performed in the landfill, wetland and ponds at the site and the reference location.

3.1.11.3.1 Landfill

Tissue samples collected from the landfill included 6 small mammals and plants consisting of house mice (*Mus musculus*), and ruderal vegetation such as ripgut brom (*Bromus diandrus*), foxtail grass (*Hordeum murinum leporinum*), and mustard (*Brassica nigra*). The small mammal and plant samples were analyzed for metals, PAHs, SVOCs, PCBs, pesticides, organotins and moisture content. Lipid analyses also were performed for small mammals. A single composite sample of invertebrates including beetles, bees, moths, butterflies, and spiders was collected from the landfill and analyzed for metals, organotins and moisture content.

3.1.11.3.2 Wetland

During the 1998 TtEMI and SOMA Ecological Risk Assessment investigation, samples were collected from six locations at the West Beach Landfill Wetland, W01 through W06. Two sampling locations, W02 and W06, were considered to be located in the wetland habitat while the remaining were located in the upland habitat. Upland plant samples consisted of wild oats (*Avena fatua*), foxtail grass (*Hordeum murinum/emporium*), lotus (Lotus sp.), and wild radish (*raphanus sativus and R. raphanistrum*). House mice samples also were collected from both the wetland and upland area. A composite sample of invertebrates was collected from each area (wetland and upland) due to the small presence at individual sampling locations. Samples were from the West Beach Landfill Wetland were analyzed for metals, PAHs, organotins, pesticides, and PCBs. The mice samples also were analyzed for moisture content and percent lipids.

Three of the six tissue sampling areas at the wetland (W03, W05, and W06) were intended to be located in upland habitat with the remaining three located in the wetland habitat. However, based on vegetation types identified during the sampling effort, only two areas (W02 and W06) were considered wetland habitat. Three upland plant samples consisting of wild oats (*Avena fatua*), foxtail grass (*Hordeum murinum/eporinum*), lotus (*Lotus* sp.), and wild radish (*Raphanus sativus* and *R raphanistrum*) were collected from the wetland. Four wetland plant samples consisting of pickleweed (*Salicornia virginica*) were collected. Upland and wetland plant samples collected from the wetland were analyzed for metals, PAHs, SVOCs, PCBs, pesticides, organotins, moisture content and percent solids. Six small mammal samples consisting of mice (*Mus musculus*) and mouse hides were collected and analyzed for metals,

PAHs, SVOCs, PCBs, pesticides, organotins, moisture content and lipids. One composite sample of miscellaneous invertebrates was collected from all the upland areas and all the wetland areas because the number of invertebrates present in each sampling area was insufficient to constitute an individual sample. The invertebrate samples were analyzed for the same parameters as small mammal tissues.

3.1.11.3.3 Ponds

The North Pond in the wetland was randomly sampled for submerged vegetation (algae) and fish by sweeping the pond with a net. The dominant fish collected were larval silversides (Family Atherinae) that were generally less than 0.8 inches (2 cm) in length. A total of 6 fish and 2 crab samples were collected from the North Pond by PRC in 1996 and analyzed for metals, PAHs, SVOCs, PCBs, pesticides and organotins. In 1998, TtEMI collected another fish sample from the North Pond and analyzed for metals, PAHs, SVOCs, PCBs, pesticides, organotins, moisture content and lipids. A single aquatic plant sample was collected from the North Pond by TtEMI in 1998 and analyzed for metals, PAHs, SVOCs, PCBs, pesticides, organotins and moisture content. Sufficient aquatic invertebrates were not found in the North Pond surface water or sediment to constitute a complete sample for chemical analyses.

One aquatic invertebrate and two terrestrial invertebrate samples were collected from the South Pond. Invertebrate samples consisted solely of backswimmers (Corixidae), wich were collected by skimming the top 4 inches (10 cm) of the South Pond with a 0.04-inch (1-mm) nylon mesh dipnet. The invertebrate samples were analyzed for metals, PAHs, SVOCs, PCBs, pesticides, organotins, moisture content, and lipids. One fish sample was collected from the South Pond and consisted of the threespine stickleback (*Gasterosteus aculeatus*), which ranged from 0.8 to 1.6 inches (2 to 4 cm) in length.

3.1.12 Reference/Background Conditions

Background studies were performed by PRC (1997b), Terranext (1998), and TtEMI (1998b) to establish the organic and inorganic background soil and groundwater concentrations at Alameda Point. In December 2001, TtEMI issued the *Summary of Background Concentrations in Soil and Groundwater* to summarize this work and the basis used to establish background conditions. In general, accepted statistical methodology was used to calculate background concentrations of inorganic and organic constituents in soil and groundwater at Alameda Point. For the purposes of this RI, only the inorganic background concentrations have been used for comparison purposes and to put site data into proper context. In 2004, the Navy performed a characterization study to establish background activity for radium 226 (Ra-226) in Alameda Point soils as part of the radiological surveying activities at IR Sites 1 and 2. This study and the results are summarized below; for additional details refer to the *Final IR Site 2 Radiological Characterization Survey Report* (TTFW, 2005). The following subsections provide additional details pertaining to the calculation of inorganic background levels in soil and groundwater.

3.1.12.1 Soil

Establishing inorganic background concentrations in soils at Alameda Point consisted of three main steps:

- Division of the installation into areas with geologically similar soils that could be represented by a single site-specific background dataset.

- Review of the Installation Restoration Program (IRP) database to identify data points that would be representative of site-specific background.

- Performance of statistical analyses to determine site-specific background concentrations.

Based on a preliminary evaluation of iron and manganese data from soils, the installation was divided into three site-specific background areas: Fill Area 1 (historically identified and currently known as the "pink area") encompassed the runway area and central portions of the installation; Fill Area 2 (historically identified and currently known as the "blue area") included the southeastern portion of the installation; and Fill Area 3 (historically identified and currently known as the "yellow area") covered the far western portion of the installation, in the area of IR Site 2. The IRP database was reviewed to select sampling data for each area. Any sample that was collected as part of a CERCLA investigation and sampling data that contained non-PAH organic chemicals were excluded. Based on these criteria, a total of 247 potential samples were selected as suitable background data for the three areas. The 80th percent lower confidence limit (LCL) of the 95th percentile of the concentration distribution (80 LCL/95) was calculated for each inorganic chemical using the methodologies described in the *Final Statistical Methodology for Background Comparisons* document (PRC, 1997b). Table 3-13 presents the inorganic background data that were established for the soils specifically in the "yellow area", as presented in *Summary of Background Concentrations in Soil and Groundwater* (TtEMI, 2001).

Background activity for Ra-226 was determined by TTFW in 2004 by selecting three separate reference areas (i.e., not the same reference areas assessed during the inorganic background concentration evaluation) located a minimum of 200 ft outside any environmentally impacted area at Alameda Point. The locations selected had not been impacted by site activities and were therefore representative of background levels. The reference areas were located upgradient and cross-gradient from IR Site 2 and physically separated from the site. Nine static background measurements and one field duplicate were obtained from each reference area to establish the background surface radiation count and exposure rate using 2-in by 2-in sodium iodide (NaI) detectors and an energy-compensated G-M detector that were subsequently used for a radiological surveying program at IR Sites 1 and 2. In addition, nine soil samples and one field duplicate were collected from 0 to 20 in bgs at each reference area to establish the background level of Ra-226 in the soil. The average background measurements for the NaI detectors and the energy-compensated G-M detector meter were 4,803 counts per minute (cpm) and 9.7 microrhoentgen per hour (μR/h). The average Ra-226 concentration measured in the reference areas was 0.365 picocuries per gram (pCi/g), which has been used as an appropriate background concentration of Ra-226 in soils at Alameda Point.

3.1.12.2 Groundwater

Inorganic background concentrations in FWBZ groundwater also were established for Alameda Point. The following methodology was used to determine potential data points that could be used in the background statistical analysis (TtEMI, 2001):

- Select monitoring wells for background concentrations that appear to be unaffected by CERCLA site-related contamination.

- Compare all organic groundwater data from the initial data set to the 1996 tap water preliminary remediation goals (PRGs) to exclude impacted wells.

- Exclude any outlier data points by using probability plots and Rosner's test.

- Test the remaining data (without outliers) for normality using a statistical graphics program.

- Prepare summary statistics and estimate the background concentration of metals in shallow groundwater.

Using this process, 35 monitoring wells were used in the statistical analysis to determine background concentrations for metals in groundwater. All data that was used were obtained from monitoring wells in the FWBZ. The SWBZ is characterized by extensive saltwater intrusion, which can prevent analytical methods from detecting trace level metals.

The same statistical procedure that was used to calculate background soil concentrations also was used to calculate background groundwater concentrations. Table 3-14 presents the inorganic background results for shallow groundwater as presented in *Summary of Background Concentrations in Soil and Groundwater* (TtEMI, 2001).

4.0: REMEDIAL INVESTIGATION METHODOLOGY AND DATA COLLECTION

4.1 Introduction and Purpose

As discussed in Section 3.0, several phases of investigative work historically have been completed at IR Site 2. Those investigations included collection and chemical evaluation of soil, sediment, groundwater, surface water, and porewater samples; assessment of ecological resources; delineation of wetlands; surveys to locate OEW and radiological material; geotechnical and seismic evaluations; and other activities. Currently, quarterly groundwater monitoring activities are ongoing using a network of monitoring wells positioned around the periphery of the site and along the boundary between the landfill and wetland portions of the site. Soil-gas sampling also is ongoing at regular intervals using a limited network of soil-gas monitoring wells within the landfill. Despite these historic and ongoing activities, until the implementation of this RI, a thorough and comprehensive characterization of all environmental media for all potential constituents of concern had not been completed that would be sufficient to fully characterize the nature and extent of contamination, develop thorough human health and ecological risk assessments for IR Site 2, and evaluate potentially necessary remedial alternatives. Specifically, the historical environmental investigations completed at the site have generally focused on the landfill and wetland boundaries, and not on the interior areas of these portions of the site.

This section discusses the general investigative scope of work that was implemented during the RI at IR Site 2 to address data gaps from previous investigations and to generate a repository of site-specific data related to the overall nature and extent of contamination. In addition, reference/background data was generated during the RI to augment existing background datasets that are used to distinguish the nature and extent of contamination at IR Site 2 from ambient conditions. In general, data were generated from areas within the footprint of the landfill and within the inundated and non-inundated portions of the wetlands, and from suitable reference/background locations (i.e., locations with characteristics similar to the site but not affected by site activities or contamination) both at Alameda Point and at a suitable site located outside the boundaries of Alameda Point. Data were acquired during two seasonal sampling events to address potential seasonal variability in some environmental media (e.g., surface water in the wetland ponds), to allow minor data gaps from the first seasonal sampling event to be filled, and to allow appropriate planning for certain events (e.g., tissue sampling) to be fully developed. The data generated during the RI according to the methodology discussed in this section are used in this RI Report to characterize the nature and extent of contamination at IR Site 2 (Section 5.0) and to fully develop human health and ecological risk assessments for the site (Sections 6.0 and 7.0).

The following subsections discuss and briefly describe the field sampling methods that were employed at IR Site 2 during the RI to generate the necessary site-specific and reference/background data. The field program implemented at IR Site 2 and at reference/background locations during the RI is summarized in significantly greater detail in Appendix B of this RI Report. The field sampling plan was developed to appropriately characterize portions of the site that had not been adequately addressed prior to this RI. Sampling locations were selected to provide reasonable and appropriate spatial coverage of the site and to address specific areas that were considered potential waste disposal areas, or on the basis of data reviewed following execution of the first seasonal sampling effort. The analytical targets evaluated at each sampling location were selected to provide necessary data given the reported site history, the nature and results of historical sampling activities, the project DQOs, data from the first seasonal sampling effort, and certain specific regulatory requirements.

Specific sampling procedures, protocols, and specific analytical methodologies related to the phases of investigation presented in the sections below have previously been described in greater detail in the Sampling and Analysis Plan (SAP), which was included as Appendix A of the Work Plan (Battelle et al., 2005). All field sampling activities were completed in strict accordance with the Site Health and Safety

Plan (SHSP), which was included as Appendix B of the Work Plan (Battelle et al., 2005). Throughout the implementation of the RI field sampling activities, all standards of care were exercised to ensure that natural resources (i.e., biota and abiotic resources) were protected to the maximum extent possible while still ensuring completion of the field sampling program and generation of the necessary data.

4.2 Preliminary Data Acquisition

Certain data were required to evaluate current site conditions and to help guide the precise implementation of the RI sampling activities. These data were collected specifically to refine project conceptual site models (CSMs) and to locate potential hazards and local anomalies that could have impacted the RI field investigations. These activities were completed prior to the dry and wet season field sampling activities and/or in conjunction with the seasonal field sampling activities.

4.2.1 Surface Water Field Measurements

Water quality measurements (including water depth, salinity, temperature, dissolved oxygen, turbidity, and pH) were collected from each wetland pond on a continuous basis for approximately eight months beginning in July 2004. These measurements were collected to characterize the basic physicochemical characteristics of the pond habitats and to evaluate general water quality and seasonal variability in the wetland ponds. Measurements were collected using a YSI Model 6600-EDS-M(S) multi-parameter water quality analyzers with data logging capability at two discrete locations in each the North Pond and South Pond. The surface water field measurement program is described in greater detail in Appendix B of this RI Report.

4.2.2 Geophysical Survey

A comprehensive geophysical survey was completed at IR Site 2 prior to implementation of the invasive RI sampling program. The geophysical survey consisted of digital geophysical mapping (DGM) using state-of-the-art time-domain electromagnetics (TDEM). The survey was implemented to locate potential buried contamination sources or obstructions (e.g., drums, containers, or oil sumps), to provide information necessary to complete invasive sampling in optimal locations, and to provide the maximum protection possible to site workers against exposure to potential subsurface hazards, including potential exposure to OEW, during sampling. The methods applied during implementation of the geophysical survey are discussed in detail in Appendix B of this RI Report.

4.2.3 Radiological Survey

At roughly the same time that the RI field sampling activities described in this RI Report were implemented, a comprehensive radiological survey program was implemented at IR Site 2 by the Navy and in accordance with a separate schedule and work plan that incorporated guidance from the Multi-Agency Radiation Survey and Site Investigation Manual (MARSSIM) (TTFW, 2005). The primary purpose of the radiological survey was to characterize the extent and depth (to a maximum of approximately 20 inches) of Ra-226 contamination potentially present at IR Site 2. Measurements and methods included surface activity scans using either a vehicle-mounted or portable sodium iodide detector, fixed gamma and exposure rate surveys, gamma energy analysis using a portable high-purity Germanium (HPGe) system, and soil sampling. Prior to conducting the radiological survey, a separate geophysical survey was completed along with land surveying and vegetation clearance. The radiological surveying activities did not extend into the wetland portion of the site to protect this sensitive habitat (i.e., some degree of dewatering reportedly would have been required to properly implement the radiation survey, which would likely have negatively impacted the wetlands).

The primary detection criterion for the radiological screening project at IR Site 2 was radium concentrations indistinguishable from reference/background for radium. As such, the radiological survey program also involved the identification of three separate reference areas on Alameda Point that were considered unaffected by site contamination. Reference areas were evaluated for background radiation levels using a sodium iodide detector and by collecting shallow (i.e., 0 to 20 inches bgs) soil samples for analysis of radium and strontium.

Over three million sodium iodide measurements were generated during the surface scanning activities at IR Site 2. Locations at IR Site 2 where surface activity was determined to exceed approximately two times the expected background count rate (i.e., a net of 4,000 counts per minute [cpm] developed through characterization at the three Alameda Point reference areas) were marked for possible focused soil sampling. Soil samples from the 23 locations at IR Site 2 exhibiting the highest survey results were evaluated in the field for gamma energy using a portable gamma energy analyzer. Subsequently, a total of six composite soil samples (including one duplicate sample) were collected from 0 to 20 inches bgs at IR Site 2 locations where radiological surveying data indicated Ra-226 levels were above background, and the samples were submitted to a laboratory for analysis of Ra-226. The soil samples were also analyzed for Strontium-90 (Sr-90). Soil sampling was conducted with direct-push drilling equipment.

Because the radiological survey was conducted by a separate contractor and was not proposed directly in the Work Plan as part of the RI, it is not described in further detail in Appendix B. However, the results of the radiological survey are summarized in Section 5.0 of this RI Report and can be reviewed in detail by referring to the *Final Radiological Characterization Survey Report* (TTFW, 2005). During the RI field sampling program specifically memorialized in this RI Report, soil sampling for radiological characterization was completed and augments the information collected during the radiological survey program (see Section 4.3).

4.3 Dry Season and Wet Season Field Sampling

To develop a fully representative dataset of environmental conditions at IR Site 2 during the RI, additional characterization was required for soil, groundwater, and tissue in the landfill portion of the site, and for soil, groundwater, sediment, surface water, and tissue in the wetland portion of the site. Sampling of these environmental media was conducted during two sampling events. One round of sampling was completed during the dry season of 2004 (October 2004) and a second round of sampling was completed during the wet season of 2005 (March 2005). During the dry season, it was easier to collect surface and subsurface soil because water levels were at or near their annual low. During the wet season, the use of the site by ecological receptors was at or near its peak, and it was anticipated that biological tissues were more easily obtained to support the ERA. The wet season also was the appropriate time to conduct necessary toxicity and bioaccumulation assays, to collect additional data to assess temporal variability in certain environmental media (i.e., surface water in the wetland ponds) with respect to the dry season data, and to collect additional soil and groundwater samples to fill minor data gaps identified in the dry season dataset. A brief description of the dry and wet season sampling events is provided below. A more detailed description of the RI field activities and figures depicting specific sampling locations are provided in Appendix B to this RI Report.

As described in Section 4.2.1 and Appendix B, the geophysical survey was conducted prior to invasive field sampling. The geophysical survey characterized potential subsurface waste disposal areas, allowing for proper placement of invasive sampling locations. The geophysical survey provided the maximum amount of protection possible for site workers against potential exposure to any subsurface hazard, including OEW. In addition, routine air monitoring was conducted during all invasive site activities in accordance with the project SHSP (i.e., Appendix B of the Work Plan). Health and safety air monitoring included regular assessment of the potential presence of volatile organic vapors and radiation using a

properly calibrated photoionization detector (PID) and a portable radiation meter, respectively. The PID and field radiation meter were also used to screen soil cores collected from the site, as detailed in Sections 4.3.1 and 4.3.2.

4.3.1 Dry Season Sampling

During the dry season sampling event, soil cores and groundwater samples were collected from the landfill and wetland areas, and surface water and sediment samples were collected from both the North and South Ponds in the wetlands. Samples were analyzed for a suite of target analytes previously detected at the site, potentially present based on existing site history information, and/or specifically requested by the regulatory agencies. A Trimble Geoexplorer® GeoXT™ hand-held global positioning system unit was used during field activities to track the coordinates of sampling locations. The global positioning system unit provided sufficient accuracy and allowed particular sampling locations to be resampled as necessary.

During the dry season field mobilization, seasonal wetlands potentially present in the northern and northeastern portions of the landfill area (see Section 2.0) were assessed. Because the document summarizing the existence of these seasonal wetlands (TTFW, 2004) was generated immediately prior to field mobilization, the condition of these areas needed to be evaluated before developing a specific sampling plan for them. It was determined that additional sampling was not required to specifically address these potentially discrete seasonal wetland areas, and that the sampling to be conducted throughout the landfill and wetlands was appropriate to characterize the site.

4.3.1.1 Dry Season Landfill and Wetland Area Soil and Groundwater Sampling

Soil

During the dry season sampling event, soil coring was completed in and around the interior portions of the landfill and interior margin, and surface soil grab sampling and soil coring were completed in the wetlands. At each coring location, a boring was advanced using direct-push (i.e., Geoprobe®) drilling techniques. Soils were retrieved and screened for volatile organic vapors using a properly calibrated PID and for radiation using a field radiation meter. Soil samples were collected from discrete sampling intervals in each core, with subsurface soil samples collected from above the water table at intervals exhibiting the greatest potential for contamination based on visual observations and/or PID screening results.

Analyses completed on soil samples collected from the landfill and wetland areas during the dry season included some or all of the following analytical parameters: PCBs; pesticides; metals; VOCs; SVOCs (including PAHs); TPH; TOC; grain-size distribution; moisture content; hexavalent chromium; explosive constituents; polychlorinated dibenzodioxins (PCDD)/polychlorinated dibenzofurans (PCDF); radio-isotopes (gross alpha, gross beta, Ra-226, Ra-228, Pb-210, and isotopic uranium [U-234, U-235 and U-238]); and tributyltin (TBT).

Appendix B of this RI Report provides a detailed summary of specific sampling methods, sampling locations, discrete sampling depths, and the analytical parameters evaluated for various soil samples collected from the landfill and wetland areas during the dry season.

Groundwater

During the dry season sampling event, groundwater samples were collected from across the landfill and wetlands. The sampling locations were biased towards areas of suspected contamination (e.g., potential drum disposal areas and oil sumps). Geoprobe® drilling techniques were used to advance to an appro-

priate depth and temporary monitoring wells were set. Groundwater samples were collected and analyzed for some or all of the following compounds: PCBs; pesticides; metals; VOCs; SVOCs (including PAHs and 1,4-dioxane); TPH; radiological parameters (gross alpha, gross beta, Ra-226, Ra-228, Pb-210, isotopic uranium, and tritium); hexavalent chromium; and explosive constituents.

Two sets of groundwater samples were collected at each sampling location. One was analyzed unfiltered and the second was filtered at the laboratory prior to analysis. The filtered and unfiltered data were necessary to allow turbidity effects to be evaluated, as the direct-push groundwater sampling produced a significant amount of sample turbidity, and elevated sample turbidity can artificially elevate groundwater concentrations of constituents that strongly sorb to soil (e.g., PCBs and metals). Filtration was conducted only for analyses for which filtration was appropriate (metals, PCBs, pesticides, SVOCs, and radio-nuclides), and was specifically not conducted for VOC analyses.

Appendix B of this RI Report provides a detailed summary of specific sampling methods, sampling locations, discrete sampling depths, and the analytical parameters evaluated for various groundwater samples collected from the landfill and wetland areas during the dry season.

4.3.1.2   Dry Season Wetland Area Surface Water and Sediment Sampling

Surface Water

During the dry season sampling event, surface water samples were collected from the North Pond and the South Pond. Samples were collected for analysis of some or all of the following analytical parameters: PCBs; pesticides; metals; VOCs; SVOCs (including PAHs and 1,4-dioxane); alkalinity; hardness; hexa-valent chromium; TPH; and radionuclides (gross alpha, gross beta, Ra-226, Ra-228, Pb-210, and isotopic uranium). As with groundwater samples collected during the dry season sampling event, two sets of surface water samples were collected at each sampling location. One was analyzed unfiltered and the second was filtered at the laboratory prior to analysis to address elevated turbidity typical of a shallow water body. Filtration was conducted only for analyses for which filtration was appropriate (metals, PCBs, pesticides, SVOCs, and radionuclides), and was specifically not conducted for VOC analyses.

Appendix B of this RI Report provides a detailed summary of specific sampling methods, sampling locations, discrete sampling depths, and the analytical parameters evaluated for various surface water samples collected from the ponds during the dry season.

Sediment

Sediment cores were collected from discrete locations in the North Pond and the South Pond during the dry season sampling event. The sediment cores were completed in the area around the perimeter of the wetland ponds that was not inundated (but showed evidence of being seasonally inundated). Sediment samples were collected from several discrete depth intervals in each core, and were collected after all of the surface water sampling was completed to minimize potential effects to the surface water from sedi-ment sampling activities. The sediment samples then were analyzed for some or all of the following analytical parameters: PCBs; pesticides; metals; VOCs; SVOCs (including PAHs); hexavalent chromium; and TPH.

Appendix B of this RI Report provides a detailed summary of specific sampling methods, sampling locations, discrete sampling depths, and the analytical parameters evaluated for various sediment samples collected from the ponds during the dry season.

4.3.2 Wet Season Sampling

Based on a preliminary review of the data available from the dry season field sampling prior to implementing the wet season field activities, additional soil and groundwater data were collected from the landfill during the wet season to fill minor data gaps and to supplement the dry season field sampling dataset. The supplemental sampling was not intended to constitute a delineation effort. Exploratory trenches also were completed during the wet season sampling event, and soil samples were collected from the trenches to further augment the site characterization dataset. Also during the wet season sampling event, additional sediment and surface water samples were collected from the wetland portion of the site to further investigate the nature and extent of contamination in this area and to allow an evaluation of potential seasonal variability in surface water contaminant concentrations and site hydrology.

In addition, the wet season sampling activities included sampling of various tissues across the landfill and wetland portions of the site. Tissue samples were collected from the landfill and wetland portions of the site during the wet season because ecological resources were more abundant and organism populations/ communities more representative compared to the dry season. Additional sampling of abiotic media (i.e., sediment and surface water) was completed during the wet season for the specific purpose of completing toxicity and bioaccumulation testing in support of the ERA.

4.3.2.1 Wet Season Landfill and Wetland Area Soil, Groundwater, and Tissue Sampling

Based on a preliminary review of the dry season field sampling data available prior to implementing the wet season sampling event, limited soil sampling was conducted in the landfill and wetland portions of the site and a very limited amount of additional groundwater sampling was conducted in the landfill area during the wet season sampling event. No additional groundwater sampling was conducted in the wetland portion of the site during the wet season, and only a limited amount of soil sampling was performed to support bioaccumulation testing as described in Section 4.3.2.4. Tissue sampling was also conducted in the landfill and wetland portions of the site during the wet season.

Soil

A limited number of additional soil cores were completed in the landfill portion of the site during the wet season sampling event. At each soil coring location, a boring was advanced using Geoprobe® drilling techniques. Soils were retrieved and screened for volatile organic vapors using a properly calibrated PID and for radiation using a field radiation meter. Soil samples were collected from discrete sampling intervals in each core for chemical analysis. Soil samples from each soil core were analyzed for a varying list of chemical targets intended to reflect the principal contaminants detected in nearby soil cores completed during the dry season field sampling program. In general, one or more of the following were analyzed in each soil sample collected from the supplemental wet season soil cores: metals; PCBs; PAHs; radionuclides; and PCDD/PCDF.

Appendix B of this RI Report provides a detailed summary of specific sampling methods, sampling locations, discrete sampling depths, and the analytical parameters evaluated for various soil samples collected from the landfill and wetland areas during the wet season.

Groundwater

A very limited number of additional hydropunch groundwater sampling locations were completed in the landfill area during the wet season sampling event. Geoprobe® drilling techniques were used to advance to an appropriate depth and a temporary monitoring well was set. Groundwater samples were collected using a peristaltic pump and tubing and analyzed for a varying list of chemical targets intended to reflect

the principal contaminants detected in nearby temporary groundwater wells and/or soil cores completed during the dry season field sampling program. Groundwater samples were analyzed for some or all of the following analytical parameters: PCBs, pesticides, metals, VOCs, SVOCs (including PAHs and 1,4-dioxane), radiological constituents (gross alpha, gross beta, Ra-226, Ra-228, Pb-210, isotopic uranium, and tritium), and explosive constituents. As with dry season groundwater sampling, two sets of groundwater samples were collected. One was analyzed unfiltered and the second was filtered at the laboratory prior to analysis to allow turbidity effects to be evaluated. Filtration was conducted only for analyses for which filtration is appropriate (metals, PCBs, pesticides, SVOCs, and radionuclides), and specifically was not conducted for VOC analyses.

Appendix B of this RI Report provides a detailed summary of specific sampling methods, sampling locations, discrete sampling depths, and the analytical parameters evaluated for the groundwater samples collected from the landfill area during the wet season.

Tissue

During the wet season sampling event, various types of invertebrates and plants were collected from the landfill and wetlands. Tissue samples were analyzed for some or all of the following analytical parameters: PCBs; pesticides; metals; and SVOCs (including PAHs). Tissue samples were collected to the extent possible from locations where samples of environmental media (e.g., surface water, sediment, or soil) also were collected. In general, sufficient plant tissue was collected for chemical analysis from most of the planned sampling locations in the landfill and wetlands. Only limited invertebrate tissue could be collected from the landfill and non-inundated portion of the wetland area, which did not allow for chemical analysis of this tissue type. Only one field mouse was collected during sampling activities at IR Site 2, and no fish or benthic invertebrates were obtained in the wetland ponds, despite reasonable efforts to collect these tissue types.

Appendix B of this RI Report provides a detailed summary of specific sampling methods, sampling locations, and the analytical parameters evaluated for various tissue samples collected from the landfill and wetland areas during the wet season. In addition, Appendix B contains a detailed discussion of those tissue sampling efforts that did not yield any tissue samples (or a sufficient volume) for chemical analysis.

4.3.2.2   Wet Season Wetland Area Surface Water and Sediment Sampling

Additional surface water and sediment sampling was conducted in the wetland ponds during the wet season sampling activities to further investigate the nature and extent of contamination in this area and to allow an evaluation of potential seasonal variability in contaminant concentrations and site hydrology.

Surface Water

Surface water samples were collected again during the wet season sampling event. Surface water was collected from the North Pond and South Pond, and all efforts were made to precisely replicate the dry season sampling locations. Two additional surface water samples were collected from the South Pond in the wet season because they could not be collected in the dry season due to the South Pond being dry in those locations. Samples were collected for analysis of some or all of the following parameters: PCBs; pesticides; metals; VOCs; SVOCs (including PAHs and 1,4-dioxane); alkalinity; hardness; hexavalent chromium; TPH; and radionuclides (gross alpha, gross beta, Ra-226, Ra-228, Pb-210, and isotopic uranium). Each surface water sample was analyzed for the same analytical targets as were evaluated for the same sampling location during the dry season sampling event. Sufficient surface water volume also was collected to ensure that the necessary toxicity testing (see Section 4.3.2.4) could be implemented.

As with the dry season sampling, two sets of surface water samples were collected at each sampling location. One was analyzed unfiltered and the second was filtered at the laboratory prior to analysis to assess turbidity-related effects. Filtration was conducted only for analyses for which filtration was appropriate (metals, PCBs, pesticides, SVOCs, and radionuclides), and specifically was not conducted for VOC analyses.

Appendix B of this RI Report provides a detailed summary of specific sampling methods, sampling locations, discrete sampling depths, and the analytical parameters evaluated for various surface water samples collected from the ponds during the wet season.

Sediment

Sediment samples were collected from discrete locations in the North Pond and South Pond during the wet season sampling event. Unlike the dry season sediment sampling, the wet season sediment sampling focused on the surface sediments beneath the water column of the wetland ponds, with the sediment sampling locations coinciding with surface water sampling locations. As with the dry season sampling, sediment sampling was completed after surface water sampling to minimize potential effects to the surface water from sediment sampling activities. Sediment samples were collected for analysis of some or all of the following analytical parameters: PCBs; pesticides; metals; VOCs; SVOCs (including PAHs); TOC; interstitial salinity; grain-size distribution; sulfides; hexavalent chromium; TPH; and radionuclides (gross alpha, gross beta, Ra-226, Ra-228, Pb-210, and isotopic uranium). Sufficient sediment volume was also collected to attempt to harvest benthic invertebrate tissue and to complete necessary toxicity and bioaccumulation testing (see Section 4.3.2.4).

Appendix B of this RI Report provides a detailed summary of specific sampling methods, sampling locations, discrete sampling depths, and the analytical parameters evaluated for various sediment samples collected from the ponds during the wet season.

4.3.2.3   Wet Season Exploratory Trenches

A limited number of exploratory trenches were excavated at the site during the wet season, focusing on areas considered likely to contain significant volumes of waste material based on geophysical survey results, analytical chemistry results from the dry season field sampling or historical sampling efforts, historical aerial photography, and/or historical information on site-specific disposal practices. The purpose of the trenches was to characterize the nature, type, and condition of the waste disposed at the site, and to determine the typical depth of waste placement.

All of the trenches were completed in the landfill area of the site. In some cases, the trenches were completed in locations coinciding with discrete waste disposal areas summarized in historical site information (see Section 2.0). Because the geophysical survey and dry season sampling data did not suggest the presence of significant waste material in the wetlands, and given the highly saturated conditions present in the wetlands during the wet season, no trenches were completed in the wetland portion of the site.

Soil samples were collected from each trench location and analyzed for varying suites of chemical analytes consistent with the principal constituents detected at nearby soil coring locations during the dry season field sampling. Trench soil samples were analyzed for some or all of the following analytical parameters: metals; PCBs; PCDD/PCDF; PAHs; and radionuclides (gross alpha, gross beta, Ra-226, Ra-228, Pb-210, and isotopic uranium).

Appendix B of this RI Report provides a detailed summary of specific trenching methods, trench soil sampling methods, trench and trench soil sampling locations, discrete sampling depths, and the analytical parameters evaluated for various soil samples collected from the trenches during the wet season.

4.3.2.4 Wet Season Toxicity and Bioaccumulation Testing

As described above, additional abiotic samples were collected at the site during the wet season to perform necessary toxicity and bioaccumulation tests in support of the ERA. The following subsections describe the tests performed. Appendix C of this RI Report provides a detailed summary of the specific toxicity and bioaccumulation testing protocols.

Toxicity Testing

Two types of acute toxicity tests were conducted on sample material from the North and South Ponds: whole sediment acute toxicity testing; and a water column toxicity testing. Each of the tests conducted during the RI is briefly described below:

- **Bulk Sediment Toxicity:** This test was conducted as a laboratory determination of acute toxicity resulting from a 10-day static exposure of the amphipod *Eohaustorius estuarius* to marine sediment. North and South Pond sediments, reference site sediments, and control sediment were included in the amphipod test.

- **Water Toxicity:** The water-column toxicity testing involved laboratory procedures used to evaluate survival and growth of invertebrate larval forms and juvenile fish. North and South Pond water samples were used as well as a reference site and control water samples. Pond water samples were inoculated for a 7-day exposure period with the mysid *Americamysis bahia* and the fish *Atherinops affinis* under controlled laboratory conditions.

Bioaccumulation Testing

The bioavailability of sediment contaminants and potential risk to upper trophic level receptors was evaluated through a laboratory bioaccumulation test to evaluate the uptake of chemicals from site sediment into the tissue of organisms. The bioaccumulation test consisted of a 28-day flow-through exposure using the bent-nose clam, *Macoma nasuta*, and the polychaete worm, *Nephtys caecoides*. The bioaccumulation testing was completed using sediment collected from the North and South Ponds, as well as a reference sampling site and a control location. In addition, bioaccumulation testing was completed using a limited number of soil samples that were collected from the wetland and a reference sampling site in the wet season. These bioaccumulations tests were done using soils because only limited site-specific tissues (e.g., terrestrial invertebrate and small mammal) could be collected from the non-inundated portions of the wetland, as described in Appendix B.

4.3.2.5 Wet Season Reference Area Sampling

Samples were collected at China Camp State Park (CCSP) and locations at Alameda Point during the wet season sampling event to generate critical reference/background data for the RI, as described below. Appendix B of this RI Report provides a detailed summary of specific sampling methods, sampling locations, discrete sampling depths, and the analytical parameters evaluated for various reference/background area samples collected during the wet season.

Soil

During the wet season sampling event, soil samples were collected from several locations at CCSP representing both upland and wetland habitat. Samples were collected using hand sampling equipment

and analyzed for some or all of the following analytical parameters: PCBs; pesticides; metals; VOCs; SVOCs (including PAHs); hexavalent chromium; PCDD/PCDF; TBT; and TPH. Additional wetland soil volume was collected from CCSP to serve as the reference exposure for bioaccumulation testing conducted during the wet season.

In addition to soil sampling at CCSP, soil samples were collected for PCDD/PCDF analysis from the same Alameda Point reference areas that were used to collect reference/background radiological data for the radiation surveying activities recently conducted by the Navy (see Section 4.2.3) (TTFW, 2005). In general, the locations of the reference areas at Alameda Point were selected to be a minimum of 200 ft outside of any impacted area associated with an Operable Unit or active site at Alameda Point. The reference areas exhibited similar physical, chemical, geological, radiological, and biological characteristics as IR Site 2, but have not been impacted by site activities.

Sediment

Sediment samples were collected from several locations at CCSP during the wet season, in both wetland depressions and channels. Sediment samples were collected and analyzed for some or all of the following analytical parameters: PCBs; pesticides; metals; VOCs; SVOCs (including PAHs); hexavalent chromium; TPH; TOC; grain-size distribution; sulfides; and interstitial salinity. Additional sediment volume was also collected to serve as the reference exposure during toxicity and bioaccumulation testing conducted during the wet season (see Section 4.3.2.4).

Surface Water

Surface water samples were collected from the same locations where sediment samplers were collected from CCSP during the wet season. Surface water samples were collected and analyzed for some or all of the following analytical parameters: PCBs; pesticides; metals; VOCs; SVOCs (including PAHs and 1,4-dioxane); hexavalent chromium; TPH; alkalinity; and hardness. In addition, general water quality parameters were measured during surface water reference sampling. Additional surface water volume was also collected to serve as the reference exposure during toxicity testing conducted during the wet season (see Section 4.3.2.4).

Tissue

Samples of plant and invertebrate tissue were collected from discrete upland and wetland locations at CCSP during the wet season. Plant tissue samples were analyzed at the laboratory for some or all of the following analytical parameters: PCBs; pesticides; metals; and SVOCs (including PAHs). Invertebrate tissues were not submitted for chemical analysis because there was insufficient volume of these same tissues collected from IR Site 2 for chemical analysis. To the extent possible, tissue samples were collected from the same locations where soil samples were collected for bioaccumulation testing.

5.0: RI RESULTS AND EVALUATION OF NATURE AND EXTENT OF CONTAMINATION

This section describes the results of the environmental quality evaluations conducted at the site during the RI. This section also summarizes the chemical constituents present in various environmental media at IR Site 2, evaluates the spatial distribution of these constituents, and evaluates the presence and spatial extent of chemical constituents in ambient environmental media at CCSP as well as other reference and/or background datasets. Data from the site are evaluated in and of themselves, and within the context of reference, or ambient, and Alameda Point background conditions to develop conclusions regarding the nature and extent of contamination at the site that can be related to historical Navy operations or that is more likely associated with ambient environmental conditions unrelated to site activities. Where appropriate and possible, site data also are evaluated relative to historically generated site data and on the basis of more refined comparison criteria, such as depth interval and season, to elucidate more detailed conclusions regarding the nature and extent of site contamination.

5.1 Methods and Data Preparation

Analytical data generated through off-site laboratory analysis of the multitude of samples collected at IR Site 2 and the project reference areas were validated in accordance with the protocols described in the project SAP (i.e., Appendix A of the Work Plan [Battelle et al., 2005]). The data and validation results were reviewed and found to meet the DQOs established for the RI. The data now available representing the various environmental media at IR Site 2 are sufficient in both quantity and quality to evaluate the overall nature and extent of contamination at IR Site 2, and to perform the human health and ecological risk assessments (see Sections 6.0 and 7.0 of this RI Report, respectively).

Prior to or in conjunction with evaluating the IR Site 2 data to assess the nature and extent of contamination at the site, the data required preparation, compilation, and graphical display. Conclusions regarding the nature and extent of contamination at IR Site 2 are based on the comparison of site-specific data versus available ambient/background information and the identified patterns, gradients, and/or trends, if any, in contaminant occurrence and distribution. General guidance exists that describes the manner in which data should be evaluated to characterize the nature and extent of contamination at a site, and this guidance was generally followed. Given the magnitude of the data generation effort of this RI, the assessment of the nature and extent of contamination also was conducted in accordance with standard industry practices and is based largely on the professional judgment and opinions of qualified professionals.

Analytical data from the various environmental media at IR Site 2 were compared to ambient/background data collected in conjunction with the RI or available from other sources. Data from IR Site 2 also were compared to relevant compound-specific benchmarks as necessary to develop an understanding of contamination patterns and significance. General data statistics were computed for the various datasets available from the RI sampling, and appropriate statistical comparisons were completed between various datasets, including historical data, RI data, and ambient/background data. Prior to applying any statistical method to a dataset, the data were evaluated to determine whether the assumptions inherent to the method were met by the data. *Guidance for Environmental Background Analysis for Soil* (U.S. Navy, 2002) and other similar guidance were consulted to select statistical methods valid for comparing site data (current and historical where appropriate) to ambient/background data and one another. Overall, all soil, groundwater, surface water, sediment, tissue, and survey data were reviewed simultaneously to develop a comprehensive conceptual model of the nature and extent of contamination at IR Site 2.

In developing the comprehensive CSM of the nature and extent of contamination at the site, the following datasets were relied on:

- **Historical Data:** All available data for the various environmental media at IR Site 2 that were generated during sampling efforts prior to the sampling conducted in direct conjunction with this RI were compiled and used as a basis of comparison to data generated specifically through the RI. The various historical sampling programs implemented at the site are described in Section 3.0 of this RI Report. Usable data were not necessarily available from each and every historical sampling event, nor were data for all media necessarily generated during each historical event. However, all data that were available and usable were compiled to form historical datasets for the various media of interest. Available historical data are collectively defined (generally by medium) as the "historical dataset" in describing their applicability for the following media.

- **Soil:** A preliminary analysis was completed by developing box and whisker plots of the method detection limits (MDLs) for various analytical targets in the historical soil dataset versus the soil data generated through implementation of this RI. Based on this graphical statistical evaluation, in virtually all cases (i.e., 123 of 129), the mean MDL associated with the newly generated RI soil data was more sensitive than the mean MDL in the historical dataset. In addition, box and whisker plots were developed to graphically display the concentrations of various analytical targets from the site based on the historical soil dataset and the more recent RI soil dataset. Based on this evaluation, there was no obvious indication that either dataset depicted a significantly greater or lesser degree of environmental impact in site soils. Furthermore, the recently generated RI soil data represent sample collection efforts focused on portions of the site where contaminant impacts would be expected to be most significant (i.e., the central portions of the site, rather than the site periphery and berms as with the historical soil data). For these reasons, the RI soil data were considered the most appropriate and suitable to comprehensively evaluate the nature and extent of contamination in soil at the site (and to develop appropriate human health and ecological risk assessments for exposures related to site soil). As such, the data used to develop the CSM of the nature and extent of contamination in soils at the site was exclusively the data recently generated during implementation of the RI methodology described in Section 4.0 of this RI Report. The RI data represent both surface and subsurface soils from the dry and wet season sampling events.

- **Sediment:** As with soil data, a preliminary evaluation of MDLs and a secondary evaluation of analyte concentrations for site sediments were completed using box and whisker plots. Based on these graphical statistical evaluations, in virtually all cases (i.e., 122 of 127), the mean MDL associated with the newly generated RI sediment data was more sensitive than the mean MDL in the historical dataset, and there was no obvious indication that either dataset depicted a significantly greater or lesser degree of environmental impact in site sediment. The RI sediment data represent a thorough spatial and temporal characterization of the sediment medium at the site. For these reasons, the recently generated RI sediment data were considered the most appropriate and suitable to comprehensively evaluate the nature and extent of contamination in sediment at the site (and to develop appropriate human health and ecological risk assessments for exposures related to site sediment). As such, the data used to develop the CSM of the nature and extent of contamination in sediment at the site was exclusively the data recently generated during implementation of the RI methodology described in Section 4.0 of this

RI Report. The RI data represent both surface and subsurface sediment from the dry season and surface sediment from the wet season.

- **Surface Water:** Aqueous media are considerably more transient than soil or sediment. As such, the most current data are generally most appropriate to develop conclusions about the nature and extent of contamination in aqueous media. The RI surface water data represent a very current and thorough spatial and temporal characterization of the surface water medium at the site. Therefore, the data used to develop the CSM of the nature and extent of contamination in surface water at the site (and to develop appropriate human health and ecological risk assessments for exposures related to site surface water) was exclusively the data recently generated during implementation of the RI methodology described in Section 4.0 of this RI Report. The RI data represent surface water samples from both the dry season and the wet season.

- **Groundwater:** As indicated for surface water, given the more transient nature of aqueous media, the most current data are generally more appropriate to develop conclusions about the nature and extent of contamination. The RI groundwater data represent a very current and thorough spatial characterization of the groundwater medium at the site and also represent sample collection efforts focused on portions of the site where contaminant impacts would be expected to be most significant (i.e., the central portions of the site rather than only the site periphery). In addition, valuable groundwater data are generated on a regular basis during an ongoing quarterly groundwater monitoring program implemented at the site by the Navy. The ongoing quarterly groundwater monitoring program focuses on wells that are located along the site periphery and the berm between the landfill and wetlands. A quarterly groundwater monitoring event completed through the ongoing monitoring program (i.e., winter 2004) coincides very closely in time with the dry season sampling event of the RI, when the majority of RI groundwater sampling was completed. Together, the RI groundwater sampling and the winter 2004 quarterly groundwater sampling event represent a very thorough spatial characterization of groundwater at the site. Therefore, the data used to develop the CSM of the nature and extent of contamination in groundwater at the site (and to develop appropriate human health and ecological risk assessments for exposures related to site groundwater) were the data recently generated during implementation of the RI methodology described in Section 4.0 of this RI Report and the data derived during the winter 2004 quarterly groundwater monitoring event implemented separately by the Navy (ITSI, 2005). The RI groundwater data were combined for evaluation specifically with groundwater data from wells in the FWBZ from the quarterly monitoring dataset. Groundwater in the SWBZ was evaluated using exclusively data from the quarterly monitoring dataset.

- **Soil Gas:** Soil gas data were not generated specifically during implementation of the RI. However, in conjunction with the ongoing quarterly groundwater monitoring program implemented at the site by the Navy, soil gas samples are collected from a network of soil gas monitoring wells in the landfill portion of the site. A quarterly soil gas monitoring event completed through the ongoing monitoring program (i.e., winter 2004) coincides very closely in time with the dry season sampling event of the RI. Therefore, the data used to develop the CSM of the nature and extent of contamination in soil gas at the site (and to develop appropriate human health and ecological risk assessments for exposures related to site soil gas) was the data derived during the winter 2004 quarterly groundwater and soil gas monitoring event implemented separately by the Navy (ITSI, 2005).

- **Tissue:** Tissue data were generated historically and during the RI at the site. However, tissue data are not generally used to describe the nature and extent of contamination at a site, but rather to support the application of appropriate risk assessment methods, such as developing appropriate uptake and/or bioaccumulation models. As such, tissue data are not discussed specifically in this section to describe the nature and extent of contamination at IR Site 2, but are discussed in detail in the development of the ERA (see Section 7.0 of this RI Report).

- **Ambient Data:** As described in Section 4.0 of this RI Report, data were collected during the RI from CCSP to serve as an ambient dataset. CCSP was selected as an appropriate reference sampling area, in close consultation with the regulatory agencies, given its geographic setting (i.e., near the site but not immediately near any industrial or commercial activity) and its historical use as a reference sampling area for other environmental investigations. Specifically, soil, sediment, surface water, and tissue samples were collected from CCSP to allow comparison between site and suitable ambient information. However, as noted during the reference sampling area selection process with regulators, developing a truly accurate ambient dataset for IR Site 2 is complicated. Ultimately, although the data generated at CCSP are considered reasonable and appropriate for developing conclusions regarding site conditions relative to ambient environmental impacts, additional effort may be required in the future (i.e., at the FS and/or remedial design phase) to fully resolve the degree of environmental impact at the site relative to ambient contributions.

The RI data for IR Site 2 are provided in Microsoft® Excel files contained on CD-ROM in Appendix D. Although tissue data are not specifically evaluated in this section, as discussed above, they are provided in Appendix D. The following specific data preparation techniques were used in assembling the data for the evaluation of the nature and extent of contamination:

- **Field Duplicates:** Prior to generating summary statistics of the various datasets, laboratory results for field duplicates were first averaged to generate an arithmetic mean value for an individual sampling point. This eliminated the potential to bias the statistics by incorporating two individual data points for one specific sampling location. Subsequently, the summary statistics largely were used to numerically describe the nature and extent of site contamination, along with visual interpretation of graphical representations of the data (see below). It should be noted that, with respect to the interpretation of bubble plots relative to summary statistics, the bubble plots provide separate data points for duplicate sample results rather than an average. Therefore, although these two tools summarize the same data, minor differences that might appear to be discrepancies may exist between the two.

- **Individual and Total PCBs and PCB Requantification:** Two separate methods were used to generate analytical PCB data. One method quantitated the seven most commonly analyzed individual PCB Aroclors (see Table 5-1), and the other quantitated 22 individual PCB congeners (see Table 5-2). Initially, total PCBs were estimated in each sample as twice the sum of 18 individual congener concentrations rather than as the sum of PCB Aroclors (Table 5-2 also identifies the individual PCB congeners that were included in the total PCB calculation). This method is consistent with guidance from the National Oceanic and Atmospheric Administration (NOAA) National Status and Trends (NS&T) program, which has been found to be accurate and reliable in calculating total PCBs when the PCB assemblage is dominated by more highly chlorinated Aroclors such as 1254 and 1260 (O'Connor, 1997). Historical data generated at IR Site 2 and reviewed

during the development of the Work Plan (Battelle, 2005) indicated that these more highly chlorinated Aroclors were detected predominantly in site media, further supporting the use of the NOAA NS&T method to calculate total PCBs.

Following the RI sampling activities it was discovered that the PCB Aroclor assemblages present in some samples at IR Site 2 were made up of more complex mixtures that included some significantly less chlorinated Aroclors such as 1221 and 1242. At that time the Navy communicated these unexpected results to the U.S. EPA and proposed a frequently applied and accepted quantification method that would identify and conservatively quantify the most dominant Aroclor in a mixture. After receiving concurrence from U.S. EPA, this method was implemented to quantify and report Aroclor concentrations in the first draft of the RI Report. Following the initial review of the RI data and preparation of the draft RI Report, discrepancies were observed between calculated total PCB concentrations using the NOAA NS&T method and individual Aroclor data. Specifically, it was observed that for certain samples collected at IR Site 2, a single Aroclor was detected at significantly greater concentration than the NOAA NS&T-derived total PCB value.

The PCB data generated at the site were evaluated closely and it was determined that no single explanation exists to understand each discrepancy in the individual versus total PCB concentrations at IR Site 2. However, it was clear that PCB composition as Aroclors and congeners was variable across the site and between different sample types, thereby adding additional complexity to the identification of the dominant Aroclor for some samples. The initial approach of identifying and quantifying the dominant Aroclor is widely used by environmental laboratories, but it was determined that improvements could be made to the accuracy and reliability of the Aroclor data given the complexities presented in IR Site 2 media. To address this issue, the original Aroclor quantification procedure was revisited and a more deliberate and comprehensive procedure was developed to accurately and conservatively quantify Aroclors in samples composed of PCB mixtures such that the sum of Aroclors provides a robust estimate of total PCBs. All PCB Aroclor data and chromatograms generated during the RI sampling activities were carefully reviewed and in some cases requantified from the raw analytical data to ensure accuracy and consistency, given the complex mixtures of PCBs detected at the site.

The description of the nature and extent of PCB contamination at IR Site 2 presented below for various media relies on the requantified PCB Aroclor data and total PCB results calculated using the NOAA NS&T method and the sum of requantified Aroclor data. When evaluating and discussing (or preparing graphics for) individual PCBs, Aroclor data have been used rather than congener data to describe PCBs in more common terminology and to minimize the number of individual figures produced. When evaluating and discussing (or preparing graphics for) total PCB results, concentrations derived both from the NOAA NS&T method and as the sum of Aroclors (requantified) have been relied on. In addition, to maintain consistency and transparency, the requantified Aroclor data (identified by the designation "MOD" following the specific Aroclor name) and the Aroclor data provided in the Draft RI Report are presented on appropriate graphics. In no instance was any PCB Aroclor data removed or replaced in this Draft Final version of the RI Report if the data were reported accurately in the Draft RI Report (i.e., in cases where only a single Aroclor was present in the sample and the original quantification method was suitable). It should be noted that, shortly before the Draft RI Report was generated, a minor analytical conversion error in a single batch of

samples was identified that affected the quantified Aroclor results for seven individual soil samples and six individual sediment samples generated at IR Site 2. Although the analytical results were updated in the project database at that time, the modified results were inadvertently not carried through either to graphics (e.g., bubble plots and plan-view figures) or to the discussion of nature and extent of PCB contamination in the initial draft of the RI Report. These are the only original Aroclor data that have been updated in this RI Report and the information provided in the remainder of this section now takes into account these corrected results.

To mitigate any potential confounding influences related to the discrepancies observed between total PCB data based on the NOAA NS&T method and the sum of Aroclors in the IR Site 2 PCB dataset, the uncertainty sections of the risk assessments for the site incorporate a risk analysis for PCBs where the sum of Aroclors is more conservative (i.e., higher) than the value derived using the NOAA NS&T method. An additional level of conservatism is incorporated into this step in that total PCB exposure point concentrations based on sum of Aroclors have been calculated using the higher of the original versus requantified Aroclor data. Overall, this approach ensures that appropriate risk management decisions can be made on the basis of reasonably conservative and unbiased risk conclusions. Sections 6.0 and 7.0 of this RI Report provide additional detail on the treatment of PCBs and the derivation of human health and ecological risk conclusions, respectively, related to PCBs in site media.

- **PCDD/PCDF Data and Toxicity Equivalents:** The method used to generate PCDD/PCDF data quantitated a list of individual dioxin and furan congeners. Prior to assessing PCDD/PCDF data, a series of steps were completed to generate an individual piece of data for each PCDD/PCDF sampling point. First, a relevant toxicity equivalence factor (TEF) was applied to each individual PCDD/PCDF congener. The 1997 World Health Organization (WHO) TEFs were used for this calculation as summarized in Table 5-3. The TEFs provide a weight to each individual PCDD/PCDF concentration in a sample relative to the most toxic congener, namely 2,3,7,8-PCDD. Second, the TEF weighted PCDD/PCDF congener concentrations for each sample were summed to generate a single toxicity equivalent (TEQ). The TEQs were subsequently used as the basis of comparison for each PCDD/PCDF data point, as they provide a consistent and toxicologically meaningful context in which to interpret the resulting concentration. Note that two TEQ values were calculated, one for humans/mammals and another for birds. In the context of characterizing the nature and extent of contamination at IR Site 2, the TEQ_{mammal} value is primarily used; however, in the human health and ecological risk assessments (Sections 6.0 and 7.0, respectively) the more appropriate TEQ value is used to calculate risks.

- **Total DDT, DDD, DDE, and DDx:** For each sample point where pesticides were evaluated, total dichlorodiphenyltrichloroethane (DDT), dichlorodiphenyldichloroethane (DDD), and dichlordiphenyldichlorethene (DDE) were calculated as the sum of the 2,4- and 4,4- isomers within each category. Total DDx for each sample point was then calculated as the sum of the total DDT, DDD, and DDE concentrations.

- **Non-Detect Values:** Individual congeners or isomers within groups of compounds (e.g., PCBs, DDTs, DDDs, and DDEs) were not included in the calculation of total concentrations if they were not detected (i.e., qualified with a "U" or "UJ"). Rather, they were set to a value of zero. In cases where all individual congeners or isomers within a compound group requiring summation were non-detect for a particular sample, the total

concentration established for the sample was non-detect with a detection limit equal to the highest detection limit of the individual congeners or isomers for the compound group of interest in that sample.

- **Plan View Concentration Plots:** To visually convey the data generated during the RI, plan view concentration plots were prepared for the various environmental media sampled at the site. Plan view data maps were prepared for various groups of analytical targets for the soil, sediment, surface water, and groundwater media. The groups of analytical targets for which plan view data maps were prepared included VOCs, PCBs and pesticides, PCDD/PCDF, metals, and SVOCs. Where pertinent, other compound classes that were analyzed in subsets of samples were grouped with similar compound classes of the broader categories above. Specifically, TBT and radionuclides were grouped with metals, TPHs were grouped with PCBs and pesticides, and explosives were grouped with VOCs. Plan view data maps were not prepared for tissue or general chemistry (e.g., TOC, alkalinity, and hardness). For the plan view concentration maps showing PCB data, both original Aroclor results and requantified Aroclor results are provided, and total PCB results are provided both according to the NOAA NS&T method and as the sum of Aroclors. The plan view concentration maps generated to support this RI Report are presented in Appendix E.

- **Reference Values and Benchmarks:** Where applicable and relevant, site data were compared against meaningful thresholds or reference values (e.g., background concentrations, ambient concentrations, or Region 9 PRGs) to facilitate the assessment of the nature and extent of contamination. The list of reference values and benchmarks was not exhaustive to complete this assessment, but was robust and appropriate. Additional benchmarks were used as appropriate and necessary during the human health and ecological risk assessments, as described in Sections 6.0 and 7.0 of this RI Report, respectively. Specifically for the purposes of evaluating the nature and extent of contamination at the site, soil data were compared against the U.S. EPA Region 9 residential PRGs issued in October 2004. Sediment data were compared against the established San Francisco Estuary Institute (SFEI) ambient sediment screening values. Aqueous data were compared against U.S. EPA Region 9 tap water PRGs and marine ambient water quality criteria (AWQC). It should be noted that the comparison between AWQC and groundwater data is highly conservative because it has not been refined by taking into account dilution mechanisms that would likely occur. Wherever available, site data also were compared against Alameda Point background values (i.e., for metals and SVOCs/PAHs) established by TtEMI (2001), and ambient data generated at CCSP. Comparisons between RI site data and the TtEMI SVOC/PAH background values (2001) are presented solely to put the site data into context, and in no way effect the SVOCs/PAHs that are evaluated in the human health and ecological risk assessments in Sections 6.0 and 7.0, respectively.

- **Bubble Plots:** To visually demonstrate the extent of potential contamination in the various environmental media at the site, bubble plots were constructed. Individual bubble plots were developed for each compound that was detected at least once in the various media sampled at the site. The media for which bubble plots were developed included surface soil (i.e., the 0 to 1 ft bgs interval), subsurface soil (i.e., the 1 to 4 ft bgs and >4 ft bgs intervals), sediment, surface water, and groundwater. Bubble plots were not developed for tissues. Sampling locations where the compound was detected are represented by bubbles, with the size and color of the bubble correlating to the magnitude of the detected concentration. Where applicable and relevant, meaningful thresholds or

reference values (e.g., background concentrations, ambient concentrations, or Region 9 PRGs) were used to define the upper and/or lower bounds of the numeric intervals correlating to bubbles of different size and/or color. For the bubble plots showing PCB data, both original Aroclor results and requantified Aroclor results are provided, and total PCB results are provided both according to the NOAA NS&T method and as the sum of Aroclors. The bubble plots generated to support this RI Report are presented in Appendix F.

- **General Summary Statistics:** To assess the various datasets used in compiling this RI Report, general summary statistics were developed. The summary stats developed for various compounds include, but are not limited to, number of samples analyzed, frequency of detection (FOD), minimum, mean, median, and maximum MDLs and detections, and 95% upper confidence limits (UCLs) of the mean. Summary statistics were developed for soil, sediment, surface water, and groundwater, and for various detailed groupings within these media (e.g., surface and subsurface soils and North Pond and South Pond). The summary statistics were developed using the SAS™ software program. The procedures used to calculate UCLs were consistent with methods and calculations that are included in U.S. EPA software, ProUCL™. Appendix G includes a more detailed discussion of the statistical calculations and the overall data management processes that were followed in preparation of this RI. Tables summarizing the summary statistics for various environmental media are provided in the subsections below.

- **Box and Whisker Plots:** To visually demonstrate the relationships between various sets of data (e.g., RI data, historical site data, and background data) and data from various portions of the site (e.g., North Pond and South Pond), box and whisker plots showing the general statistical distribution of the various data populations of interest were developed. The box and whisker plots were developed using standard methods in the SAS software program. Box and whisker plots were only developed for individual compounds that were detected at least once in a particular medium. However, non-detect values were incorporated into the plots to demonstrate detection limits. The box and whisker plots generated to support this RI Report are presented in Appendix H.

The comprehensive CSM developed for IR Site 2 on the basis of the data evaluation described and summarized in this section is conveyed in the human health and ecological risk assessments in Sections 6.0 and 7.0 respectively. Section 8.0 summarizes the exposure media, pathways and chemicals that are identified as potential risk drivers in the human health and ecological risk assessments, and also describes the environmental fate and transport of those compounds that are of primary interest on the basis of the assessment of the nature and extent of contamination discussed below, and risk assessment results.

5.2 General RI Results

The general results generated during preliminary activities implemented in support of the RI and the exploratory trenching are described below.

5.2.1 General Surface Water Quality

As described in Section 4.0 and Appendix B of this RI Report, a surface water monitoring program was implemented in the wetland ponds beginning in July 2004 and lasting for approximately eight months until March 2005. Water quality measurements (including depth, salinity, temperature, dissolved oxygen [DO], turbidity, and pH) were collected at 15 minute intervals from each wetland pond using multi-parameter water quality analyzers. These measurements were collected to characterize the basic physico-

chemical conditions of the pond habitats and to evaluate general water quality and seasonal variability in the wetland ponds. Measurements were initially recorded at one discrete location in the North Pond and two locations in the South Pond. As indicated in Appendix B, a second meter could not be deployed initially in the North Pond in July 2004 due to minimal water depth. However, a second meter was deployed in the North Pond in January 2005 after the depth of the pond had increased in response to precipitation.

For both wetland ponds, surface water temperature clearly exhibited seasonal variability in response to ambient temperatures. Surface water temperatures were relatively warm in July and August 2004 and then steadily fell through the fall and winter months for both ponds. Average daily surface water temperatures in both ponds were generally approximately 20 to 25 degrees Celsius (°C) between July and September 2004 and declined to approximately 15 to 20°C between September and early November 2004. Average daily surface water temperatures continued to fall throughout the early part of winter, and appear to have reached their lowest at approximately 10°C in January 2005. Between January and late February 2005, average daily surface water temperatures in both ponds moderated to approximately 15°C. Temperatures appear to have continued to rise in both ponds during the early spring with rising ambient temperatures. By March 2005, when measurements were concluded, average daily surface water temperatures in both ponds were approximately 15 to 20°C, similar the temperatures observed in the previous fall and early winter. Diurnal variability was also observed in surface water temperatures in both ponds. For both ponds, daily temperature fluctuations of up to 10 to 15°C were observed in the summer and early fall months (i.e., July to September 2004). Through the late fall and winter of 2004, and the winter and spring of 2005, the amplitude of diurnal surface water temperature fluctuations in the South Pond were muted, generally showing little diurnal variability. However, with the exception of January 2005, the typical amplitude of diurnal surface water temperature fluctuations remained much greater in the North Pond between late fall 2004 and spring 2005. The typically greater amplitude diurnal surface water temperature fluctuation in the North Pond could be related to lower water depth compared to the South Pond (i.e., temperature variability would be less responsive with greater depth and volume to ambient heating and cooling) as described below.

Water depth responded somewhat differently in the two ponds over the course of the data collection period. In the South Pond, water depth increased throughout the monitoring period, and was characterized by significant increases following substantial precipitation events. Surface water depth in the South Pond was roughly 1 to 2 ft at the beginning of the monitoring period (i.e., July 2004), remained relatively static until September 2004, and increased to approximately 5 ft by the end of the monitoring period (i.e., March 2005). While the overall trend was an increase in surface water depth, minor decreases in water depth were observed in the South Pond throughout the monitoring period. The overall increase in surface water depth in the South Pond was related to the occurrence of significant precipitation generally between September 2004 and January 2005, including several precipitation events of 1 to 3 in, while the periods of decreasing depth within the overall increasing trend were likely the result of surface water infiltrating the subsurface between precipitation events and establishing equilibrium within the hydrologic system. In the North Pond, water depth also responded to substantial precipitation events, but the seasonal increase was limited (or muted) and more transient in comparison to the South Pond. Surface water depth in the North Pond was roughly 1 ft in July 2004, remained relatively static through September 2004, increased to roughly 2 ft by the middle of January 2005, and declined to depths similar to the July 2004 baseline by March 2005. As with the South Pond, the overall increase in surface water depth in the North Pond between September 2004 and January 2005 was related to the occurrence of significant precipitation, including several precipitation events of 1 to 3 in. Relatively significant changes in surface water depth were observed in the North Pond in response to discrete rain events. As with the South Pond, periods of decreasing depth within the overall North Pond surface water depth trend were likely the result of surface water infiltrating the subsurface between precipitation events and establishing equilibrium within the hydrologic system. In addition, surface water depth in the North Pond was likely more erratic but muted

because the North Pond is connected to the San Francisco Bay by a 36-inch pipe culvert, which would effectively prevent even significant inputs of water from having a long-term or high magnitude impact on the pond system.

Dissolved oxygen concentrations in both ponds exhibited a strong diurnal cycle during the monitoring period, likely driven by algal oxygen production during daylight and nighttime respiration by indigenous microorganisms. In the North Pond, daytime DO concentrations varied between generally 20 and 40 mg/L, while nighttime DO concentrations fell to as far as 0 to 5 mg/L. In the South Pond, daytime DO concentrations were not as high as in the North Pond, generally varying between 10 and 25 mg/L, while nighttime DO concentrations also fell to as low as 0 to 5 mg/L. Dissolved oxygen concentrations and the amplitude of diurnal fluctuations were generally lower in both ponds during the winter months, likely the result of the reduction in photosynthesis caused by shortened periods of daylight and lower water temperatures. However, the diurnal variability in DO levels remained higher in the North Pond than the South Pond even during the winter months. This is likely related to the connection between the North Pond and San Francisco Bay, which would potentially allow microorganisms and already oxygenated surface water to enter. Alternatively, given the lack of connection between the South Pond and San Francisco Bay, biotic oxygenation and respiration would likely slow to a much greater degree in the winter. Anomalous DO measurements were observed for at least one meter in each pond (i.e., the North Pond west meter and the South Pond east meter during January 2005), likely the result of fouling of the DO sensors.

Turbidity in both ponds also appeared to exhibit a diurnal cycle. The diurnal variability in turbidity in the ponds was likely the result of several factors, including the diurnal cycling of algal growth and afternoon winds disturbing the fine sediments in the shallow ponds. Surface water turbidity also appeared to be responsive to precipitation, with turbidity levels generally increasing sharply following the input of turbid overland runoff and then sharply decreasing with the settling out of suspended material between precipitation events. In both ponds, turbidity varied diurnally from nearly 0 nephalometric turbidity units (NTU) to as high as 1,200 NTU. In the North Pond, the diurnal variability in turbidity appeared throughout the monitoring period, while in the South Pond, turbidity generally remained significantly lower between the late fall of 2004 and spring of 2005 (i.e., After November 2004, turbidity in the South Pond generally remained below 200 NTU). This is potentially explained by the connection between the North Pond and San Francisco Bay, which would allow bay surface water with a potentially elevated suspended solids load to enter the pond over the course of typical tidal cycling even during winter. In addition, biotic activity was likely more suppressed in the South Pond compared to the North Pond over the winter, potentially contributing to overall lower turbidity measurements (i.e., related to biomass) in the South Pond during the winter months.

In the North Pond, surface water pH appeared to remain relatively stable throughout the monitoring period, generally fluctuating between approximately 7.5 and 9.5 standard units (S.U.). Surface water pH did appear to rise slightly in the North Pond between July 2004 and December 2004, then fall slightly between December 2004 and March 2005. However, the overall trend in the North Pond was highly static. At the South Pond western meter, surface water pH appeared to decrease relatively steadily between July 2004 and January 2005, and then rise between late winter and early spring 2005. As with the North Pond, the fluctuation in surface water pH at the South Pond western meter was generally between 7.5 and 9.5 S.U. At the South Pond eastern meter, surface water pH dropped steadily throughout the monitoring period, beginning at approximately 9 S.U. and dropping to approximately 7 S.U. by the end of the survey period. A diurnal fluctuation in surface water pH was observed during all seasons and at all monitoring stations. Diurnal variability was likely linked to the consumption of CO_2 by algae during daylight photosynthesis and the production of CO_2 during dark respiration. The magnitude of the observed diurnal fluctuation was generally lower in the South Pond (i.e., generally less than 0.2 S.U.) compared to the North Pond (i.e., generally 0.5 to 1 S.U.). The reason for the pH variability between meter stations in the South Pond is not readily explained.

Surface water salinity in the North Pond appeared to increase from approximately 30 ppt at the beginning of the monitoring period to approximately 90 ppt in the late fall of 2004, and then decrease from that level to approximately 10 ppt by the end of the monitoring period. This same general trend was observed for the eastern meter station in the South Pond, where the initial surface water salinity was approximately 70 ppt, increased to approximately 140 ppt by the late fall of 2004, then decreased to approximately 110 ppt by the end of the monitoring period. This pattern is likely due to initial dissolution of salts from dry pond sediments inundated with the onset of rainy season conditions, followed by dilution of salinity impacts with successive precipitation inputs. For the South Pond western meter station, salinity appeared to decrease throughout the monitoring period from an initial level of approximately 60 ppt to a final level of approximately 15 ppt. The reason for the salinity variability between meter stations in the South Pond is not easily explained, but could be the result of the sediments at the western meter location containing lower concentrations of salts or probe malfunction at the western meter station.

Figures presented in Appendix I show the temperature, water depth, salinity, DO, turbidity, and pH data generated at the meter stations in the wetland ponds over the course of the water quality monitoring investigation.

5.2.2 Geophysical Surveying

A comprehensive geophysical survey was completed at IR Site 2 prior to implementation of the invasive RI sampling program. The geophysical survey is discussed in Section 4.0 and in greater detail in Appendix B of this RI Report.

The geophysical survey consisted of DGM using state-of-the-art TDEM equipment. The survey was implemented to locate potential buried contamination sources or obstructions (e.g., drums, containers or oil sumps), to provide information necessary to complete invasive sampling in optimal locations, and to provide the maximum protection possible to site workers against exposure to potential subsurface hazards, including potential exposure to OEW, during sampling.

The geophysical survey was successful in identifying the presence and general extent of subsurface geophysical anomalies. The output of the geophysical survey was data representative of the magnitude of electromagnetic response relative to a calculated background level. Specific buried wastes could not be resolved or described, but general locations and magnitudes of likely buried debris areas were mapped. In general, no geophysical anomalies were identified in the wetland area. A small anomaly in the far south-western corner of the wetlands could represent historical scrap metal disposal (see Section 2.0), but the identified anomaly was highly localized and the particular waste type could not be determined. Through-out the landfill and southeastern portion of the interior margin, extensive areas of relatively high magnitude electromagnetic response were detected, indicating a highly diffuse pattern of historical waste disposal. Several localized areas of elevated electromagnetic response also were detected in the western portions of the interior margin. These findings are entirely consistent with known or suspected site history (see Section 2.0). The geophysical survey could not, in and of itself, specifically confirm or refute the existence of the specific historical waste disposal areas described in Section 2.0.

Appendix B of this RI Report provides the final Power Surveying geophysical survey report and a draw-ing showing the final geophysical survey data generated at the site. The colors displayed on the drawing in Appendix B correlate to the magnitude of electromagnetic return, with green representing very low electromagnetic return (or background), blue representing a moderate degree of electromagnetic return, and red representing a significant electromagnetic anomaly.

After the geophysical survey was completed, the proposed locations of all subsurface borings and explor-atory trenches completed during the RI were assessed with respect to the electromagnetic anomaly data

produced during the DGM. Using the accurate location data generated along with the electromagnetic anomaly data during the geophysical survey, all boring locations to be completed during the RI were assigned more precise physical coordinates corresponding to surface locations with a detected electromagnetic response consistent with background. All invasive boring activities were subsequently successful, and no significant issues related to potentially dangerous subsurface obstructions were encountered in the field.

5.2.3 Radiological Surveying

As discussed in Section 4.2.3, a comprehensive radiological survey program was implemented by the Navy at IR Site 2 at roughly the same time as the RI field activities (TTFW, 2005). The radiological survey was implemented in accordance with a separate schedule and work plan. Measurements and methods included surface activity scans using either a vehicle-mounted or portable sodium iodide detector, fixed gamma and exposure rate surveys, gamma energy analysis using a portable HPGe system, and soil sampling.

The radiological survey program involved the identification of three separate reference areas on Alameda Point that were considered unaffected by site contamination. Reference areas were evaluated for background radiation levels using a sodium iodide detector and a HPGe system and by collecting shallow soil samples (i.e., 0 to 20 in bgs) for analysis of radium. Nine individual sodium-iodide and HPGe measurements were recorded at each reference location. Shallow soil samples for analysis of Ra-226 were then collected at each sodium-iodide/HPGe measurement location. The mean Ra-226 concetrations in the reference areas were 0.309 pCi/g, 0.379 pCi/g, and 0.407 pCi/g, and the overall average background concentration of Ra-226 detected in the Alameda Point reference areas was 0.365 pCi/g. This background value has been used as a general reference value in evaluating the nature and extent of contamination at IR Site 2 (see Section 5.3). However, because this average background value is based on a limited amount of surface soil sampling completed during a surface radiation characterization survey of IR Sites 1 and 2, its applicability to all of IR Site 2 is inherently complicated. Nevertheless, the background value developed during the radiological survey program represents a reasonable point of general comparison for IR Site 2 RI data.

More than 3 million individual sodium iodide detector measurements were collected during the radiological survey of IR Site 2, representing 992,594 unique geographic coordinates. Sodium-iodide activities calculated during the radiological survey for the majority of the site generally were consistent with background measurements generated at the Alameda Point reference areas. Several discrete and localized portions of IR Site 2 exhibited sodium iodide activities higher than background. Specifically, 891 of the 992,594 sodium-iodide measurements at IR Site 2 demonstrated activities above reference. These 891 survey locations were grouped by geographic location into 23 discrete and localized areas at IR Site 2, primarily along the northern and eastern sides of IR Site 2 and in the vicinity of the bunkers in the interior margin.

The 23 localized areas at IR Site 2 with sodium-iodide activities greater than background were further evaluated using in situ gamma spectroscopy, which indicated the presence of Ra-226 at each location. Soil samples were evaluated in the field for gamma energy using a portable gamma energy analyzer, and six soil samples (including one duplicate) were subsequently collected from IR Site 2 for laboratory analysis of Ra-226. The detected Ra-226 concentrations in four of the five soil samples collected at the site were marginally above the background value determined from the Alameda Point reference area sampling. The maximum concentration of Ra-226 detected in IR Site 2 soil was 0.397 pCi/g compared to the average background value of 0.365 pCi/g.

As indicated in Section 4.2.3, soil samples collected from IR Site 2 and reference locations were also analyzed for Sr-90. This radionuclide was not detected at concentrations greater than method detection limits in reference area soil samples, and was only detected in one IR Site 2 soil sample at a very low concentration of 0.214 pCi/g.

Detailed results of the 2004 radiological survey can be reviewed in detail by referring to the *Final Radiological Characterization Survey Report* (TTFW, 2005). A graphical presentation of the data generated from the 2004 radiological survey is provided on Figure 5-1.

Because the Ra-226 data generated during the radiological survey (TTFW, 2005) was very limited relative to the data generated during the RI, and because the concentrations of Ra-226 detected during the RI are generally higher than the concentrations observed in the 2004 radiological survey soil sampling dataset (and therefore provide a more conservative and worse case assessment of the site), the Ra-226 data from the 2004 radiological survey were not incorporated into the summary statistics, bubble plots, box and whisker plots, or overall discussion of radionuclides at the site. In addition, Sr-90 was not specifically evaluated during the RI, and there is no reason to believe this compound represents a concern at the site.

5.2.4 Trenching

Five exploratory trenches were excavated at the site during the wet season, focusing on areas considered likely to contain significant volumes of waste material based on geophysical survey results, analytical chemistry results form the dry season field sampling or historical sampling efforts, historical aerial photography, and/or historical information on site-specific disposal practices. The purpose of the trenches was to characterize the nature, type, and condition of the waste disposed at the site, and to determine the typical depth of waste placement.

Because the possibility existed for uncovering OEW, qualified OEW technicians from Power Surveying were present to monitor the trenching areas prior to and during excavation activities. Each trench was excavated to the water table using a backhoe and was approximately 3 ft wide (i.e., the approximate width of an excavator bucket) and 25 ft long. High-intensity artificial lights were used to illuminate the sidewalls of the open trenches, and both still photographs and video were taken to document excavation activities at each trench. Waste debris was first encountered at varying depths, generally between 1.5 ft and 3.5 ft bgs. A wide variety of waste and debris was encountered during the process and included glass, plastic (e.g., sheeting and toys), metal (e.g., posts, sheet metal, and siding), wood, canvas, paper, concrete, rubber (e.g., tire and hose), cable, boots, Styrofoam, carpeting, fabric, film, microfiche, and a newspaper dated 15 December 1975. No OEW, drums, cylinders, radiological waste, or other potentially hazardous materials were identified during the trenching operations. Videography conducted during the trenching activities is provided in Appendix B.

5.3 Nature and Extent of Contamination

All soil, groundwater, surface water, sediment, tissue, and survey data were reviewed to develop a comprehensive CSM of the nature and extent of contamination at IR Site 2. Before developing the CSM, each available dataset generated at the site during the RI was assessed with respect to other existing site data from the same medium (e.g., soil, surface water, sediment, or groundwater) to determine which data were most suitable and appropriate to describe environmental conditions at IR Site 2.

The nature and extent of contamination in each environmental media is described in detail below, separated by landfill and wetland portions of the site.

5.3.1 Landfill

The following subsections describe the most likely sources of contamination in the landfill portion of the site, and describe the comprehensive CSM of site contamination developed for the landfill portion of the site by environmental medium (i.e., surface and subsurface soil, groundwater, and soil gas) and analyte class (e.g., metals, VOCs, and pesticides).

5.3.1.1 Potential Sources of Contamination

As indicated in Section 2.0, historical information suggests that limited waste disposal activities may have occurred in the far northeastern portion of the site during the early 1950s, and that disposal activities began in earnest at the landfill only after completion of the sea wall in 1956. Waste was initially placed starting in the northern portion of the landfill, eventually extending to most of the northern and eastern areas of the landfill. It is estimated that the landfill received a maximum of 1.6 million tons of general base garbage (i.e., general household waste, food waste, paper products, etc.) over the course of its operation as the main disposal location for Alameda Point.

The following specific wastes were reportedly disposed of at the landfill over the course of its operation (E&E, 1983): waste chemical drums; solvents; oily waste and sludge; paint waste; plating wastes; industrial strippers and cleaners; acids; mercury; PCB-containing liquids; batteries; low-level radiological waste from radium dials and dial paints; scrap metal; inert ordnance; asbestos; several pesticides (solid and liquid); tear gas agent; biological waste (i.e., potentially infectious waste from Oak Noll Naval Hospital and laboratory waste from the Naval Supply Center at Oakland); creosote; dredge spoils; and waste medicines and reagents. Specific disposal activities that were identified through an evaluation of facility records and/or interviews with NAS personnel during the IAS (E&E, 1983) include:

- Asbestos pipe lagging removed from ships was placed in central and eastern parts of the landfill;

- Pesticides were disposed of in the southeastern portion of the landfill;

- Roads on the landfill were constructed in part from spent sandblasting grit and abrasives used to rework metal parts and strip paint from ship hulls;

- Waste oils and PCB-containing oils were used as dust suppressants on the roads in the northeastern corner of the landfill;

- Several hundred pounds of tear gas agents were disposed in containers as a loose powder at the landfill in 1968 or 1969;

- Mercury waste from manometers and fluorescent light bulbs was drummed and disposed at the landfill between 1952 and 1968;

- Two unlined pits in the landfill (one located in the northeastern corner of the landfill and one in the mid-western portion of the site, immediately south of the landfill/wetland boundary) were used to contain waste oils; and

- Sludge and grit from 15 oil-water separators located at Alameda Point were disposed of at a rate of approximately 53,000 gallons of sludge per month during the 1960s.

Historical disposal methods at the site generally consisted of trench and fill operations. A trench would be excavated to the water table and progressively filled with wastes, then the waste material would be spread and compacted using heavy machinery. The landfill area was covered with cover soil on an intermittent basis. Reportedly, during its early years of operation, full drums were buried at the site. However, after three separate landfill fires were triggered by damaging full drums of material with heavy machinery, all drums were reportedly punctured and drained before being disposed (E&E, 1983).

To aid in identifying potential contamination source areas (i.e., hot spots) within the landfill, historical information from the IAS (E&E, 1983) was used to develop a map identifying discrete wastes that were disposed at IR Site 2. The locations of these discrete waste disposal areas were mapped during the IAS. The discrete disposal areas are listed below, and are summarized in greater detail in Section 2.11 and shown on Figure 2-10.

- **Asbestos Disposal Areas:** Asbestos pipe lagging historically removed from ships was reportedly disposed of in the mid-eastern section of the landfill.

- **OEW Burial Site:** An approximately 2.5-acre area known as the Possible OEW Burial Site is located in the southeast corner of the landfill.

- **Unlined Oil Pits:** Two unlined pits were reportedly located at the landfill for waste oils that were not reclaimed or sold.

- **Radioactive Waste Storage Shack:** A former radioactive waste storage shack was located in the northern portion of the landfill.

- **PCB-Containing Oil:** Oil from transformers that potentially contained PCBs was reportedly spread on the roadways in the northeastern corner of the landfill.

- **Landfill Roadways:** Roads on the landfill were reportedly constructed in part from spent sandblasting grit and abrasives from refinishing of metal parts and blasting of ship hulls. It also was reportedly a common practice to spread oily material onto the roads at the site for dust suppression by directly discharging from moving tanker trucks.

- **Chemical Drum Disposal Areas:** Disposal of chemical drums took place in four areas reportedly termed the "chemical dump".

- **Pesticide Disposal Area:** In the late 1970s, the Public Works Pest Control Shop (i.e., IR Site 8) removed its inventory of pesticides subject to TSCA and FIFRA, and disposed of these pesticides in the landfill.

As indicated in Section 5.2.2, the geophysical survey implemented during the RI was successful in identifying the presence and general extent of subsurface geophysical anomalies. Throughout the landfill and southeastern portion of the interior margin, extensive areas of relatively high magnitude electromagnetic response were detected, indicating a highly diffuse pattern of historical waste disposal. Several localized areas of elevated electromagnetic response also were detected in the western portions of the interior margin. These findings are entirely consistent with known or suspected site history. The geophysical survey could not, in and of itself, specifically confirm or refute the existence of the specific historical waste disposal areas described above and in Section 2.0.

In addition, as indicated in Section 5.2.4, a wide variety of waste and debris was encountered during the exploratory trenching completed during the RI, including glass, plastic (e.g., sheeting and toys), metal

(e.g., posts, sheet metal, and siding), wood, canvas, paper, concrete, rubber (e.g., tire and hose), cable, boots, Styrofoam, carpeting, fabric, film, microfiche, and a newspaper dated 15 December 1975. No OEW, drums, cylinders, radiological waste, or other potentially hazardous materials were identified during the trenching operations.

5.3.1.2   Landfill Surface Soils

During the course of the RI at IR Site 2, a total of 52 surface soil samples (i.e., from the 0 to 1 ft bgs interval) were collected from the landfill area of the site. Samples were generated during the dry and wet season soil sampling programs, as well as the wet season trenching effort. The following analytical parameters were evaluated in the surface soil samples collected from the landfill area:

- All 52 surface soil samples were analyzed for metals and moisture content;

- 50 surface soil samples were analyzed for SVOCs (including PAHs);

- 44 surface soil samples were analyzed for pesticides and PCBs;

- 40 surface soil samples were analyzed for VOCs;

- 22 surface soil samples were analyzed for PCDDs/PCDFs;

- 20 surface soil samples were analyzed for hexavalent chromium;

- 19 surface soil samples were analyzed for TPHs;

- 14 surface soil samples were analyzed for Ra-226 and Ra-228;

- 10 surface soil samples were analyzed for TOC and grain-size distribution;

- 8 surface soil samples were analyzed for TBT, gross alpha, gross beta, Pb-210, U-234, U-235, and U-238; and

- 2 surface soil samples were analyzed for explosive constituents.

Table 5-4 provides a general statistical summary of the surface soil data generated in the landfill portion of the site during the RI.

Metals

Aluminum, arsenic, barium, chromium, cobalt, copper, iron, lead, magnesium, manganese, mercury, nickel, silver, vanadium, and zinc were detected in all of the surface soil samples collected from the landfill. Antimony, beryllium, selenium, and thallium were detected in 80% or more of the landfill area surface soil samples. Cadmium was detected in roughly 58% and molybdenum was detected in roughly 13% of the surface soil samples from the landfill. TBT was detected in approximately 88% of the surface soil samples analyzed for this compound, and hexavalent chromium was detected in only 5% of the surface soil samples analyzed for this metal (i.e., only one of the 20 samples analyzed for this metal).

Widespread exceedances of established Alameda Point background values for metals were identified in surface soils across the landfill for aluminum, barium, cadmium, chromium, cobalt, copper, iron, lead, magnesium, manganese, mercury, nickel, vanadium, and zinc. Antimony, arsenic, and beryllium were identified at a small number of landfill area surface soil sampling locations with detected concentrations exceeding Alameda Point background levels. TBT was detected at low concentrations at several landfill area surface soil sampling locations, with the highest detected concentration occurring in the south central portion of the landfill. When detected, hexavalent

chromium, molybdenum, selenium, silver, and thallium were only identified at very low concentrations, with none of the detected silver concentrations exceeding the applicable background value. Alameda Point background values have not been established for TBT, hexavalent chromium, molybdenum, selenium, or thallium in soils. Iron was detected at several landfill surface soil sampling locations at concentrations exceeding the Region 9 PRG, and vanadium was detected at a single sampling location in the west central portion of the landfill at a concentration exceeding the Region 9 PRG. Arsenic was detected across the landfill at concentrations exceeding the Region 9 PRG (note that the PRG for arsenic is lower than the established Alameda Point background value for this metal). The following metals were not detected above the applicable Region 9 PRG: aluminum, antimony, barium, beryllium, cadmium, chromium, cobalt, copper, hexavalent chromium, lead manganese, molybdenum, nickel, selenium, silver, thallium, and zinc (see Table 5-1).

In general, there is no obvious pattern in the occurrence of metals concentrations in landfill surface soils that would suggest localized hot spots of metals contamination or clear source areas. For lead, the maximum concentrations in surface soils appear to be located in the central portion of the landfill and the vicinity of the former radioactive waste storage shed. It is not clear, however, that this condition is the result of specific sources of lead. Maximum concentrations of manganese in landfill surface soils similarly seem to be limited to the central portion of the landfill, but it is also unclear if this condition is the result of isolated contamination sources. Maximum concentrations of mercury and zinc are limited to discrete surface soil sampling locations in the immediate center of the landfill area. The maximum concentration of TBT was detected at one surface soil sampling location in the west central portion of the landfill area, coinciding with a former landfill roadway. As described in Section 4.0, analysis for TBT was conducted only for surface soil sampling locations with the greatest likelihood to have been impacted by this compound.

Reference surface soil samples were collected from the CCSP upland for the analysis of metals. Metals concentrations detected in surface soils in the landfill portion of the site are generally consistent with metals concentrations detected in reference surface soil samples collected from the upland portion of CCSP. The mean detected concentrations of the following metals in surface soils were actually higher in the CCSP upland relative to the landfill portion of the site: aluminum; arsenic; barium; beryllium; chromium; cobalt; copper; iron; lead; magnesium; manganese; nickel; selenium; thallium; and vanadium. In addition, the maximum detected concentrations of the following metals in surface soils were actually higher in the CCSP upland relative to the landfill portion of the site: barium; beryllium; chromium; cobalt; iron; magnesium; manganese; nickel; selenium; and vanadium. Reference surface soil samples from the CCSP upland were not analyzed for hexavalent chromium or TBT.

Metals data for soil samples from the site are presented on Figure E-1 in Appendix E. Bubble plots showing the concentrations of metals in surface soil samples collected from the site during the RI are presented as Figures F-1 through F-23 in Appendix F-1. The box and whisker plots presented as Figures H-1 through H-23 in Appendix H-1 demonstrate the relationships between data associated with various metals in landfill area surface soils and metals data associated with CCSP upland reference surface soils.

SVOCs/PAHs

The following SVOCs were not detected in any surface soil samples collected from the landfill portion of the site: 2,4,5-trichlorophenol, 2,4,6-trichlorophenol; 2,4-dichlorophenol; 2.4-dimethyl-phenol; 2,4-dinitrophenol; 2-chlorophenol; 2-methyl-4,6-dinitrophenol; 2-nitrophenol; 3-nitroaniline; 4-bromophenyl phenyl ether; 4-chloro-3-methylphenol; 4-chlorophenyl phenyl ether; 4-nitroaniline; 4-nitrophenol; atrazine; *bis*(2-chloroethoxy)methane; *bis*(2-chloroethyl)ether;

bis(chloroisopropyl)ether; caprolactam; hexachlorobutadiene; hexachlorocyclopentadiene; hexachloroethane; isophorone; and *n*-nitrosodi-*n*-propylamine.

Numerous SVOCs/PAHs were detected in the surface soil samples collected from the landfill portion of the site. The following SVOCs were detected in 5% or less of the surface soil samples collected from the landfill portion of the site: 1,2,4,5-tetrachlorobenzene; 2-chloronaphthalene; 4-methylphenol; 2-nitroaniline; 3,3-dichlorobenzidine; 4-chloroaniline; diethyl phthalate; dimethyl phthalate; di-*n*-octyl phthalate; hexachlorobenzene; *n*-nitrosodiphenylamine; and pentachlorophenol. The following SVOCs were detected in roughly 6% to 16% of the surface soil samples collected from the landfill area: 1,1-biphenyl; 4-methylphenol; acetophenone; benzaldehyde; and phenol. 2-methylnaphthalene, acenaphthene, acenaphthylene, *bis*(2-ethylhexyl) phthalate, butyl benzyl phthalate, carbazole, dibenz(a,h)anthracene, dibenzofuran, di-*n*-butyl phthalate, fluorene, and naphthalene were detected in approximately 40% to 82% of the surface soil samples collected from the landfill. Anthracene, benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(g,h,i)perylene, benko(k)fluoranthene, chrysene, fluoranthene, indeno(1,2,3-cd)pyrene, phenanthrene, and pyrene were all detected in 86% to 100% of the surface soil samples collected from the landfill (see Table 5-1).

Overall, detected concentrations of SVOCs/PAHs in surface soils occur widespread over the majority of the landfill portion of the site. For certain SVOCs/PAHs, the most elevated concentrations detected in landfill area surface soils appear to correlate to sampling locations in presumed specific historical waste disposal areas, including the pesticide disposal area, drum disposal areas, and the vicinity of the former radioactive waste storage shed. However, for the most part, there does not appear to be a clear pattern of SVOC/PAH occurrence in surface soils in the landfill portion of the site. For the majority of SVOCs/PAHs for which Region 9 PRGs exist, the maximum detected analyte concentration in landfill surface soils is well below applicable PRGs. However, concentrations of benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, dibenz(a,h)anthracene, and indeno(1,2,3-cd)pyrene were detected in landfill area surface soil samples at concentrations exceeding Region 9 PRGs. Concentrations exceeding relevant PRGs also are relatively widespread in the landfill and do not appear to demonstrate a clear pattern that would suggest localized sources of SVOC/PAH contamination or contaminant hot spots.

Reference surface soil samples were collected from the CCSP upland for the analysis of SVOCs/PAHs. Although several SVOCs/PAHs were detected in the reference surface soil samples, the number of individual SVOCs/PAHs detected in the CCSP surface soils was significantly less than the number of individual SVOC/PAH compounds detected in surface soils from the landfill portion of the site. Concentrations of several SVOCs/PAHs detected in the CCSP upland reference samples were appreciably higher than concentrations detected in surface soils from the site landfill. However, for several other SVOC/PAH compounds detected in both IR Site 2 landfill area surface soils and CCSP reference surface soil samples, the maximum concentrations detected were either highly consistent or actually higher for the CCSP reference samples. Specifically, maximum concentrations of acetophenone, benzo(a)anthracene, benzo(b)fluoranthene, benzo(k)fluoranthene, and chrysene detected in surface soil samples from CCSP were highly consistent with maximum concentrations from surface soils in the landfill portion of the site, and maximum concentrations of benzo(a)pyrene, benzo(g,h,i)perylene, and indeno(1,2,3-cd)pyrene were actually higher for reference surface soil samples from CCSP compared to IR Site 2 landfill area surface soil samples. Concentrations of benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, dibenz(a,h)anthracene, and indeno(1,2,3-cd)pyrene were detected in surface soil samples from the CCSP upland at concentrations exceeding Region 9 PRGs.

There are specific background values that have been developed for certain SVOC/PAH compounds in soils at Alameda Point. For all SVOC/PAH compounds for which an Alameda Point background value has been developed (i.e., benzo(a)pyrene, benzo(g,h,i)perylene, chrysene, fluoranthene, indeno(1,2,3-cd)pyrene, phenanthrene, and pyrene), the maximum detected concentrations in surface soils from the landfill portion of the site are well below the established background level.

SVOC/PAH data for soil samples from the site are presented on Figure E-2 in Appendix E. Bubble plots showing the concentrations of SVOCs/PAHs in surface soil samples collected from the site during the RI are presented as Figures F-24 through F-64 in Appendix F-2. The box and whisker plots presented as Figures H-24 through H-66 in Appendix H-2 demonstrate the relationships between data associated with various SVOCs/PAHs in landfill area surface soils and SVOC/PAH data associated with CCSP upland reference surface soils.

PCBs

All Aroclors were detected in landfill area surface soils with the exception of Aroclor 1016. Aroclor 1221 was detected in only approximately 2% of the surface soil samples collected from the landfill area of the site (i.e., 1 of 44 samples analyzed for PCBs). The one detected concentration of Aroclor 1221 in landfill area surface soil occurred in the eastern central portion of the landfill and was approximately 8 mg/kg.

Aroclors 1232, 1242, 1248, and 1260 were detected in approximately 14% to 27% of the surface soil samples collected from the landfill area of the site and analyzed for these constituents (i.e., 27% for Aroclor 1232, 16% for Aroclor 1242, 18% for Aroclor 1248, and 14% for Aroclor 1260). The maximum detected concentrations of Aroclors 1232, 1242, 1248, and 1260 were approximately 1.7 mg/kg, 5.3 mg/kg, 0.6 mg/kg, and 0.7 mg/kg, respectively. Aroclor 1254 was detected in approximately 66% of the surface soil samples collected from the landfill. The maximum concentration of Aroclor 1254 detected was approximately 4.5 mg/kg. Landfill area surface soil sampling locations with detected concentrations of Aroclors 1232, 1242, 1248, and 1260 generally were confined to western, central, and northeastern portions of the landfill area, whereas Aroclor 1254 was detected in surface soil generally across the landfill. Higher concentrations of Aroclor 1254 also were detected in the vicinity of the former radioactive waste storage shed. Exceedances of the Region 9 PRG for Aroclor 1254 were observed at several surface soil sampling locations in the central and northern portions of the landfill and in the vicinity of the former radioactive waste storage shed.

Total PCB concentrations calculated for landfill area surface soils using the NOAA NS&T method generally were less than 1.5 mg/kg. A limited number of sampling locations in the landfill were characterized by NOAA-based total PCB concentrations of up to approximately 2.5 mg/kg, and these locations with the most significant total PCB values are confined to the central, north central, and former radioactive waste storage shack portions of the landfill/interior margin. Total PCB concentrations in surface soils calculated using the NOAA method exceed the Region 9 PRG for several locations in the landfill portion of the site, generally in the western, central, and north central portions of the landfill, in the eastern and former radioactive waste storage shack portions of the interior margin, and to a limited extent in the far southern portion of the landfill. Total PCB concentrations calculated as the sum of Aroclors show highly similar trends as the NOAA-based values. However, a number of total PCB concentrations based on the sum of Aroclors are up to 5 mg/kg, and the maximum total PCB concentration in landfill surface soil derived as the sum of Aroclors is approximately 13.3 mg/kg. The highest total PCB concentrations calculated for surface soils in the landfill, both using the NOAA method and as the sum of Aroclors, do appear to correlate to some degree to areas of presumed specific historical waste disposal, including former drum and oil disposal locations.

There are no specific background values for PCB Aroclors or total PCBs in soils at Alameda Point. Reference surface soil samples were collected from the CCSP upland for analysis of PCBs, but PCB Aroclors were not detected in these reference surface soil samples. However, total PCB concentrations were calculable for one reference surface soil sample collected at CCSP (only by the NOAA method, given that Aroclors were not detected). The total PCB concentration in this sample was significantly lower than the mean and maximum calculated total PCB concentrations for landfill area surface soils.

Aroclor and total PCB data for soil samples from the site are presented on Figure E-3 in Appendix E. Bubble plots showing the concentrations of PCB Aroclors and total PCBs in surface soil samples collected from the site during the RI are presented as Figures F-65a through F-72c in Appendix F-3. The box and whisker plots presented as Figures H-67a through H-74c in Appendix H-3 demonstrate the relationships between data for various PCB Aroclors and total PCBs in landfill area surface soils and Region 9 PRGs.

Pesticides

Endrin, endrin ketone, and toxaphene were not detected in any of the surface soil samples from the landfill area of the site that were evaluated for pesticides. The following pesticides were detected in 5% or fewer of all surface soil samples from the landfill area: *alpha*-BHC; aldrin; *beta*-BHC; *delta*-HCH; endosulfan I; endrin aldehyde; heptachlor; and methoxychlor.

2,4-DDE, endosulfan sulfate, heptachlor epoxide, and *gamma*-BHC (lindane) were detected in approximately 7% to 18% of the landfill area surface soil samples analyzed for pesticides. 2,4-DDD, 2,4-DDT, *cis*-nonachlor, dieldrin, endosulfan II, and *gamma*-chlordane were detected in approximately 50% to 85% of the surface soil samples from the landfill analyzed for pesticides. 4,4-DDD, 4,4-DDE, 4,4-DDT, *alpha*-chlordane, and *trans*-nonachlor were all detected in 90% or more of the surface soil samples from the landfill that were analyzed for pesticides.

Detections of pesticides in surface soils appear widespread across the landfill portion of the site. In general, there does not appear to be a clear pattern in the occurrence of pesticides in landfill area surface soils. In several cases, elevated levels of pesticides appear to correlate to areas of presumed historical waste disposal, including the former radioactive waste storage shed, drum disposal areas, and oil disposal areas. Furthermore, for several pesticides, the highest detected concentrations in surface soils at the landfill appear to be isolated to the former drum disposal area in the northern portion of the landfill. However, for all pesticides detected in landfill area surface soils for which there exists a Region 9 PRG, the maximum detected concentration was well below the PRG.

Reference surface soil samples were collected from the CCSP upland for the analysis of pesticides. Although several individual pesticides were detected in the reference surface soil samples from the CCSP upland, the number of pesticides detected in the CCSP surface soils was significantly less than the number of individual pesticides detected in surface soils from the landfill portion of the site. The maximum concentrations of those pesticides detected in surface soils from both the site landfill and the CCSP upland generally were appreciably lower for the CCSP soils. There are no specific background values for pesticides in soils at Alameda Point.

Pesticide data for soil samples from the site are presented on Figure E-3 in Appendix E. Bubble plots showing the concentrations of pesticides in surface soil samples collected from the site during the RI are presented as Figures F-73 through F-93 in Appendix F-4. The box and whisker plots presented as Figures H-75 through H-103 in Appendix H-4 demonstrate the relationships between data associated

with various pesticides in landfill area surface soils and pesticide data associated with CCSP upland reference surface soils.

VOCs

Only four VOCs were detected in any of the surface soil samples collected from the landfill portion of the site that were analyzed for these compounds. Acetone was detected in approximately 18% and methylene chloride was detected in roughly 53% of the landfill area surface soil samples analyzed for VOCs. The maximum detected concentrations of both of these VOCs were very low (i.e., 56 µg/kg for acetone and 140 µg/kg for methylene chloride) and both are common laboratory contaminants.

2-butanone (methyl ethyl ketone, or MEK) was detected in only one surface soil sample from the landfill at a concentration of 12 µg/kg. 4-methyl-2-pentanone (methyl isobutyl ketone, or MIBK) also was detected in only one surface soil sample from the landfill, at a concentration of 8.5 µg/kg. The detected concentrations of MEK and MIBK are well below the respective Region 9 PRGs for these compounds. Neither detection appears to represent a discrete source area or evidence of any substantial VOC contamination in surface soil in the landfill area.

Specific background values have not been developed for VOCs in soils at Alameda Point. Reference surface soil samples were collected from the CCSP upland for analysis of VOCs, but VOCs were not detected in these reference surface soil samples.

VOC data for soil samples from the site are presented on Figure E-4 in Appendix E. Bubble plots showing the concentrations of VOCs in surface soil samples collected from the site during the RI are presented as Figures F-94 through F-101 in Appendix F-5. The box and whisker plots presented as Figures H-104 through H-127 in Appendix H-5 demonstrate the relationships between data associated with various VOCs in landfill area surface soils and VOC data associated with CCSP upland reference surface soils.

PCDD/PCDF

Several individual PCDD/PCDF congeners were detected in most of the surface soil samples collected from the landfill area of the site that were analyzed for these compounds. In accordance with common practice and guidance from U.S. EPA, the individual PCDD/PCDF congeners detected in discrete surface soil samples were summed as described in Section 5.1 to generate a TEQ for each sampling point.

Overall, it appears that elevated levels of PCDDs/PCDFs are widespread in surface soils across the landfill portion of the site. For the majority of landfill area surface soil sampling locations where PCDDs/PCDFs were analyzed, the calculated TEQ exceeds the Region 9 PRG (i.e., the PRG for 2,3,7,8-TCDD, the most toxic of the PCDD/PCDF congeners). The highest calculated TEQs appear to be confined to the central portion of the landfill area, but do not appear to be collocated with presumed areas of specific historical waste disposal (i.e., drum or oil disposal areas).

Reference surface soil samples were not collected from the upland portion of CCSP for the analysis of PCDDs/PCDFs. However, specific sampling was completed during the RI to evaluate the potential presence and magnitude of PCDD/PCDF contamination at Alameda Point reference locations (i.e., locations on Alameda Point but not affected in any known way by historical industrial or waste disposal activities). Specifically, soil sampling was completed at the same Alameda Point reference locations as were assessed during the radiological survey program (see Section 5.2.3 and TTFW, 2005). Several individual PCDD/PCDF congeners were detected in most of the surface soil samples

collected from the Alameda Point reference locations. However, the concentrations of individual PCDD/PCDF congeners detected in the Alameda Point reference samples generally were lower than those detected in surface soil samples collected from the landfill portion of the site. Similarly, TEQs calculated for the sampling locations in the Alameda Point reference areas generally were lower than TEQs calculated for surface soil sampling locations in the landfill portion of the site. No calculated TEQs for reference area surface soil samples exceeded the Region 9 PRG (i.e., the PRG for 2,3,7,8-TCDD).

TEQs calculated for soil samples from the site are presented on Figure E-3 in Appendix E. Figure F-102 in Appendix F-6 is a bubble plot of TEQs generated for surface soil samples collected from the site during the RI. The box and whisker plots presented as Figures H-128 through H-146 in Appendix H-6 demonstrate the relationships between data for various PCDDs/PCDFs and TEQs in landfill area surface soils and the reference PCDD/PCDF and TEQ data developed for Alameda Point.

Petroleum Hydrocarbons

Diesel-range organics (DRO) and residual-range organics (RRO) (i.e., petroleum constituents not captured in the specific DRO or gasoline-range organics [GRO] analyses) were detected in all of the surface soil samples collected from the landfill area that were analyzed for these compounds. GRO was detected in roughly 50% of all surface soil samples collected from the landfill area that were analyzed for petroleum hydrocarbons. The maximum concentrations of DRO, GRO, and RRO detected in landfill area surface soil samples were 1,200 mg/kg, 49 mg/kg, and 10,000 mg/kg, respectively.

The highest concentrations of petroleum hydrocarbons appear to be isolated to several discrete sampling locations in the landfill area of the site. Specifically, the most elevated levels of petroleum hydrocarbons were detected in presumed areas of specific historical waste disposal, including former drum, oil, and pesticide disposal locations, as well as at one sampling location along a former landfill roadway.

Specific background values have not been developed for TPHs in soils at Alameda Point. TPHs were not analyzed for in the reference surface soil samples collected from the CCSP upland. In addition, there are no PRGs for DRO, GRO, or RRO.

TPH data for soil samples from the site are presented on Figure E-3 in Appendix E. Bubble plots showing the concentrations of petroleum hydrocarbons in surface soil samples collected from the site during the RI are presented as Figures F-103 through F-105 in Appendix F-6. The box and whisker plots presented as Figures H-147 through H-149 in Appendix H-7 demonstrate the relationships between TPH data in landfill area surface soils, subsurface soils, and soils in the wetland area of the site.

Radionuclides

Lead 210 was detected in only one of the 8 landfill area surface soil samples evaluated for this analyte, at a concentration of 0.79 pCi/g. U-235 was detected in 5 of the 8 surface soil samples analyzed for radionuclides, at a maximum concentration of 0.08 pCi/g. Gross alpha was detected in 7 of the 8 surface soil samples from the landfill that were analyzed for this analyte, at a maximum concentration of 19.1 pCi/g. Gross beta, U-234, and U-238 were all detected in all of the landfill area surface soil samples evaluated for these radionuclides, with maximum concentrations of 31 pCi/g, 0.53 pCi/G, and 0.54 pCi/g, respectively. Overall, gross alpha, gross beta, U-234, and U-238 were detected primarily in areas of presumed discrete historic waste disposal, including the vicinity of the

former radioactive waste storage shed. However, as discussed in Section 4.0, samples for analysis of radionuclides were selected primarily from locations that corresponded to areas of suspected specific historical waste disposal.

Ra-228 was detected in all of the landfill area surface soil samples analyzed for this constituent, and Ra-226 was detected in over 60% of the surface soil samples analyzed for this compound. The maximum concentration of Ra-228 detected in surface soils from the landfill portion of the site was 0.73 pCi/g. The minimum concentration of Ra-226 detected in surface soils from the landfill area of the site was approximately 0.3 pCi/g (i.e., roughly equal to the background value developed during the radiological survey described in Section 5.2.3) and the maximum concentration detected was approximately 1.8 pCi/g. The highest Ra-226 concentrations detected in landfill area surface soil samples appear to be isolated to the eastern central portion and a small area in the southern portion of the landfill.

Specific background values have not been developed for radionuclides in soils at Alameda Point other than the value generated by the Navy during its recent survey program (see Section 5.2.3). Radionuclides were not analyzed for in the reference surface soil samples collected from the CCSP upland. In addition, there are no PRGs for radionuclides.

Radionuclide data for soil samples from the site are presented on Figure E-1 in Appendix E. Bubble plots showing the concentrations of radionuclides in surface soil samples collected from the site during the RI are presented as Figures F-106 through F-113 in Appendix F-7. The box and whisker plots presented as Figures H-150 through H-157 in Appendix H-8 demonstrate the relationships between radionuclide data in landfill area surface soils, subsurface soils, and soils in the wetland area of the site.

Explosives

Only two explosive constituents were detected in surface soil samples collected from the landfill portion of the site that were analyzed for these compounds. 2,4-dinitrotoluene and 1,3-dinitrobenzene were detected in one of the two surface soil samples collected from the landfill area that were analyzed for explosive constituents. The concentrations of 2,4-dinitrotoluene and 1,3-dinitrobenzene were 0.28 mg/kg and 0.31 mg/kg, respectively, which are well below the Region 9 PRGs for these compounds. The only location where the explosive constituents were detected was in the southeastern corner of the landfill. However, this corresponds with the suspected OEW disposal area and is the only location at which surface soil samples were analyzed for explosive constituents.

No specific background values have been developed for explosives in soils at Alameda Point. Explosives were not analyzed for in the reference surface soil samples collected from the CCSP upland.

Explosives data for soil samples from the site are presented on Figure E-4 in Appendix E. Bubble plots showing the concentrations of explosives in surface soil samples collected from the site during the RI are presented as Figures F-114 and F-115 in Appendix F-8. The box and whisker plots presented as Figures H-158 through H-161 in Appendix H-9 demonstrate the relationships between explosives data in landfill area surface and subsurface soils.

General Chemistry

Several surface soil samples collected from the landfill portion of the site were evaluated for moisture content, TOC, and grain-size distribution. These analytical parameters are, in and of themselves, not

indicative of any type or distribution of contamination in landfill area surface soils, and therefore are not discussed here.

5.3.1.3   Landfill Subsurface Soils

During the course of the RI at IR Site 2, a total of 90 subsurface soil samples (i.e., from depths below 1 ft bgs) were collected from the landfill area of the site. Samples were generated during the dry and wet season soil sampling programs, as well as the wet season trenching effort. The following analytical parameters were evaluated in the subsurface soil samples collected from the landfill area:

- All 90 subsurface soil samples were analyzed for metals, SVOCs (including PAHs), and moisture content;
- 81 subsurface soil samples were analyzed for pesticides and PCBs;
- 75 subsurface soil samples were analyzed for VOCs;
- 42 subsurface soil samples were analyzed for TPHs;
- 39 subsurface soil samples were analyzed for hexavalent chromium;
- 38 subsurface soil samples were analyzed for PCDDs/PCDFs;
- 26 subsurface soil samples were analyzed for Ra-226 and Ra-228;
- 19 subsurface soil samples were analyzed for TOC and grain-size distribution;
- 16 subsurface soil samples were analyzed for TBT;
- 15 subsurface soil samples were analyzed for gross alpha, gross beta, Pb-210, U-234, U-235, and U-238;
- 4 subsurface soil samples were analyzed for explosive constituents; and
- 2 subsurface soil samples were analyzed for sulfide.

Table 5-5 provides a general statistical summary of the subsurface soil data generated in the landfill portion of the site during the RI.

Metals

Aluminum, arsenic, barium, chromium, cobalt, copper, iron, lead, magnesium, manganese, mercury, nickel, silver, vanadium, and zinc were detected in 100% of the subsurface soil samples collected from the landfill area. Antimony, beryllium, cadmium, selenium, and thallium were detected in 65% or more of the subsurface soil samples. Molybdenum was detected in roughly 33% of the subsurface soil samples, and TBT was detected in approximately 38% of the subsurface soil samples analyzed for this compound. Hexavalent chromium was not detected in any of the subsurface soil samples analyzed for this metal. In general, the frequencies of detection of individual metals in landfill area subsurface soils are highly consistent with the frequencies of detection of these metals in landfill area surface soils.

Widespread exceedances of established Alameda Point background values for metals were identified in subsurface soils across the landfill for aluminum, barium, cadmium, chromium, cobalt, copper, iron, lead, magnesium, manganese, mercury, nickel, vanadium, and zinc. In general, there is no obvious pattern in the occurrence of these metals in landfill area subsurface soils that would suggest localized hot spots of contamination or clear contaminant source areas. However, lead was detected at two subsurface sampling locations at concentrations that were markedly higher than all other

locations. The box and whisker plot for lead (provided in Appendix H-1) shows that concentrations in two subsurface sampling locations within the landfill (SOC06 and SOC21) were much higher (i.e., approximately 60,000 mg/kg) than the majority of samples.

Antimony, arsenic, beryllium, and silver are characterized by a small number of sampling locations exceeding Alameda Point background levels, with generally no evidence of localized hot spots or clear source areas. Chromium, iron, and lead were detected in several locations in subsurface soils in the landfill at concentrations exceeding Region 9 PRGs. Arsenic was detected in subsurface soils across the landfill at concentrations exceeding the Region 9 PRG (note that the PRG for arsenic is lower than the established Alameda Point background value for this metal). When detected, molybdenum, selenium, and silver were detected only at low concentrations with none of the detected silver concentrations exceeding the applicable background value and none exceeding Region 9 PRGs. TBT was detected at several sampling locations at generally low concentrations. Alameda Point background values have not been established for TBT, hexavalent chromium, molybdenum, selenium, or thallium in soils.

The mean detected concentrations of the following metals were highly consistent between landfill area subsurface soils and landfill area surface soils: aluminum, arsenic, barium, beryllium, cadmium, chromium, cobalt, iron, magnesium, manganese, mercury, molybdenum, nickel, selenium, silver, thallium, and vanadium. The mean detected concentrations of antimony, copper, lead, zinc, and TBT generally were appreciably higher in the landfill area subsurface soils compared to landfill area surface soils. The maximum detected concentrations of barium, beryllium, magnesium, nickel, selenium, thallium, and vanadium generally were consistent between landfill area subsurface and surface soils. The maximum detected concentrations of aluminum, antimony, arsenic, cadmium, chromium, cobalt, copper, iron, lead, manganese, mercury, molybdenum, silver, zinc, and TBT generally were appreciably higher in landfill area subsurface soils compared to landfill surface soils.

For some metals, the maximum concentrations in subsurface soils appear to be located in the vicinity of presumed locations of specific historical waste disposal. It is not clear, however, that this condition is the result of specific sources of metals. The maximum concentration of TBT was detected at one subsurface soil sampling location in the central portion of the landfill area, coinciding with a former landfill roadway. As described in Section 4.0, analysis for TBT was conducted only for surface soil sampling locations with the greatest likelihood to have been impacted by this compound.

Reference subsurface soil samples were not collected from the CCSP upland for the analysis of metals. However, the reference surface soil samples collected from the CCSP upland provide a valuable basis for comparison. Metals concentrations detected in subsurface soils in the landfill portion of the site are generally consistent with metals concentrations detected in reference surface soil samples collected from the upland portion of CCSP. The mean detected concentrations of the following metals were actually higher in CCSP upland surface soils relative to subsurface soils in the landfill portion of the site: aluminum; arsenic; beryllium; cobalt; iron; magnesium; manganese; nickel; selenium; thallium; and vanadium. In addition, the maximum detected concentrations of the following metals in surface soils from the CCSP upland were actually higher relative to subsurface soils from the landfill portion of the site: barium; beryllium; magnesium; manganese; nickel; selenium; and vanadium. Reference surface soil samples from the CCSP upland were not analyzed for hexavalent chromium or TBT.

Metals data for subsurface soil samples from the site are presented on Figures E-5 and E-6 in Appendix E. Bubble plots showing the concentrations of metals in subsurface soil samples collected from the site during the RI are presented as Figures F-116 through F-161 in Appendix F-9. Bubble plots were generated for both the 1 to 4 ft bgs interval and the interval below 4 ft bgs. The

description of the nature and extent of metals in landfill area subsurface soils above integrates the entire subsurface zone below 1 ft bgs. The box and whisker plots presented as Figures H-1 through H-23 in Appendix H-1 demonstrate the relationships between data for various metals in landfill area subsurface soils and metals data associated with CCSP upland reference surface soils. These box and whisker plots also demonstrate the relationships between data for various metals in landfill area subsurface soils and metals in landfill area surface soils.

SVOCs/PAHs

The following SVOCs were not detected in any subsurface soil samples collected from the landfill portion of the site: 2,4,5-trichlorophenol; 2,4,6-trichlorophenol; 2,4-dichlorophenol; 2-methyl-4,6-dinitrophenol; 2-nitroaniline; 2-nitrophenol; 3,3-dichlorobenzidine; 3-nitroaniline; 4-bromophenyl phenyl ether; 4-chloro-3-methylphenol; 4-chlorophenyl phenyl ether; 4-nitroaniline; 4-nitrophenol; atrazine; *bis*(2-chloroethoxy)methane; *bis*(2-chloroethyl)ether; *bis*(chloroisopropyl)ether; hexachlorobenzene; hexachlorobutadiene; hexachlorocyclopentadiene; hexachloroethane; isophorone; and *n*-nitrosodi-*n*-propylamine.

Numerous SVOCs/PAHs were detected in the subsurface soil samples collected from the landfill portion of the site. The following SVOCs were detected in 5% or less of the subsurface soil samples collected from the landfill portion of the site: 1,2,4,5-tetrachlorobenzene; 2,4-dimethylphenol; 2-chloronaphthalene; 2-chlorophenol; 2-methylphenol; 4-chloroaniline; dimethyl phthalate; and di-*n*-octyl phthalate. The following SVOCs were detected in roughly 9% to 19% of the subsurface soil samples collected from the landfill area: 1,1-biphenyl; 4-methylphenol; acetophenone; benzaldehyde; caprolactam; diethyl phthalate; *n*-nitrosodiphenylamine; pentachlorophenol; and phenol.

Acenaphthene, acenaphthylene, anthracene, benzo(k)fluoranthene, *bis*(2-ethylhexyl) phthalate, butyl benzyl phthalate, carbazole, dibenz(a,h)anthracene, dibenzofuran, di-*n*-butyl phthalate, fluorene, and naphthalene were detected in approximately 36% to 77% of the surface soil samples collected from the landfill. Benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(g,h,i)perylene, chrysene, fluoranthene, indeno(1,2,3-cd)pyrene, phenanthrene, and pyrene were all detected in 83% to 100% of the subsurface soil samples collected from the landfill.

Overall, detected concentrations of SVOCs/PAHs in subsurface soils occur widespread over the majority of the landfill portion of the site. For certain SVOCs/PAHs, the most elevated concentrations detected in landfill area subsurface soils appear to correlate to sampling locations in presumed specific historical waste disposal areas, including the pesticide disposal area, drum disposal areas, and the vicinity of the former radioactive waste storage shed. However, there does not appear to be a clear pattern of SVOC/PAH occurrence in subsurface soils in the landfill portion of the site that would suggest localized sources of contamination. For the majority of SVOCs/PAHs for which Region 9 PRGs exist, the maximum detected analyte concentration in landfill surface soils is well below the applicable PRG. However, concentrations of benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(k)fluoranthene, *bis*(2-ethylhexyl)phthalate, carbazole, chrysene, dibenz(a,h)anthracene, dibenzofuran, indeno(1,2,3-cd)pyrene, and naphthalene were detected in landfill area subsurface soil samples at concentrations exceeding Region 9 PRGs. Concentrations exceeding relevant PRGs also are relatively widespread in the landfill and do not appear to demonstrate a clear pattern that would suggest localized sources of contamination or contaminant hot spots.

Although the mean and/or maximum detected compound concentrations for certain SVOCs/PAHs generally were consistent between landfill area subsurface and surface soils, in most cases the mean

and maximum detected individual SVOC/PAH concentrations were appreciably higher for the landfill area subsurface soil samples compared to the landfill area surface soil samples.

Reference subsurface soil samples were not collected from the CCSP upland. However, reference surface soil samples were collected from the CCSP upland for the analysis of SVOCs/PAHs, and provide a valuable basis of comparison. Although several SVOCs/PAHs were detected in the reference surface soil samples, the number of SVOCs/PAHs detected in the CCSP surface soils was appreciably less than the number of individual SVOC/PAH compounds detected in subsurface soils from the landfill portion of the site. The maximum concentrations of all of the SVOCs/PAHs detected in both subsurface soils from the landfill and surface soils from the CCSP upland generally were appreciably lower for the CCSP soils. However, the mean concentration of acetophenone, benzo(a)pyrene, benzo(b)fluoranthene, benzo(g,h,i)perylene, benzo(k)fluoranthene, dibenz(a,h)anthracene, and indeno(1,2,3-cd)pyrene were higher for surface soil samples collected from the upland of CCSP compared to the mean concentrations of these compounds in subsurface soils from the landfill area. Concentrations of benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, dibenz(a,h)anthracene, and indeno(1,2,3-cd)pyrene also were detected in surface soil samples from the CCSP upland at concentrations exceeding Region 9 PRGs.

There are also specific background values for certain SVOC/PAH compounds in soils at Alameda Point. Background values exist for benzo(a)pyrene, benzo(g,h,i)perylene, chrysene, fluoranthene, indeno(1,2,3-cd)pyrene, phenanthrene, and pyrene. The maximum detected concentrations of these compounds in subsurface soils from the landfill portion of the site are well below the background level developed with the exception of fluoranthene, phenanthrene, and pyrene.

SVOC/PAH data for subsurface soil samples from the site are presented on Figures E-7 and E-8 in Appendix E. Bubble plots showing the concentrations of SVOCs/PAHs in subsurface soil samples collected from the site during the RI are presented as Figures F-162 through F-234 in Appendix F-10. Bubble plots were generated for both the 1 to 4 ft bgs interval and the interval below 4 ft bgs. The description of the nature and extent of SVOCs/PAHs in landfill area subsurface soils above integrates the entire subsurface zone below 1 ft bgs. The box and whisker plots presented as Figures H-24 through H-66 in Appendix H-2 demonstrate the relationships between data for various SVOCs/PAHs in landfill area subsurface soils and SVOC/PAH data associated with CCSP upland reference surface soils. These box and whisker plots presented also demonstrate the relationships between data for various SVOCs/PAHs in landfill area subsurface soils and SVOCs/PAHs in landfill area surface soils.

PCBs

As with landfill area surface soils, all Aroclors other than Aroclor 1016 were detected in landfill-area subsurface soil. Aroclors 1221, 1232, 1242, 1248, and 1260 were detected in approximately 19% to 26% of the subsurface soil samples collected in the landfill area of the site that were analyzed for these compounds (i.e., 23% for Aroclor 1221, 22% for Aroclor 1232, 26% for Aroclor 1242, 19% for Aroclor 1248, and 25% for Aroclor 1260). The maximum detected concentrations of Aroclors 1221, 1232, 1242, 1248, and 1260 were approximately 36 mg/kg, 11.9 mg/kg, 12.9 mg/kg, 3.6 mg/kg, and 46.3 mg/kg, respectively. Aroclor 1254 was detected in approximately 40% of the subsurface soil samples collected from the landfill area that were analyzed for PCBs. The maximum detected concentration of this Aroclor was approximately 7.9 mg/kg.

Subsurface soil sampling locations with detected concentrations of PCB Aroclors generally were widespread across the landfill area. However, few detections of any PCB Aroclors other than Aroclor 1260 occurred in the southern portion of the landfill. Elevated concentrations of PCB Aroclors were detected in the vicinity of presumed discrete historical waste disposal areas, including drum disposal

areas, oil disposal areas, and the former radioactive waste storage shed. Exceedances of the Region 9 PRG for Aroclor 1254 in subsurface soils were observed at several discrete locations in the landfill area, generally confined to the central portion of the landfill and the vicinity of the former radioactive waste storage shed.

Total PCB concentrations calculated for subsurface soils in the landfill portion of the site using the NOAA NS&T method exceeded 1.5 mg/kg at many sampling locations, with a maximum of approximately 52 mg/kg (i.e., at sampling location SOC11). Total PCB concentrations calculated as the sum of Aroclors show highly similar concentrations and trends as the NOAA-based values, with the maximum calculated concentration being approximately 46 mg/kg and also coinciding with sampling location SOC11. Calculated total PCB concentrations using both the NOAA method and the sum of Aroclors in subsurface soils exceed the Region 9 PRG for locations across the landfill portion of the site. In addition, the highest total PCB concentrations calculated in subsurface soils for the landfill using both the NOAA method and as the sum of Aroclors appear to correlate at least to some degree to areas of presumed specific historical waste disposal, including former drum and oil disposal locations. The highest detected total PCB concentration using both the NOAA method and as the sum of Aroclors occurred at a location specifically intended to coincide with a presumed historical drum disposal area.

Mean and maximum detected Aroclor and total PCB concentrations generally were higher in landfill area subsurface soils compared to landfill area surface soil samples. The only exception to this is the mean detected concentration of Aroclor 1221 in landfill area subsurface soil samples, which was lower than the mean detected concentration in landfill area surface soils (note there was only one detection of Aroclor 1221 in landfill area surface soil).

Specific background values have not been developed for PCB Aroclors or total PCBs in soils at Alameda Point. In addition, subsurface soil samples were not collected from the CCSP upland as part of the reference sampling program during the RI. However, surface soil samples were collected from the CCSP upland for the analysis of PCBs. Although these data would provide a valuable basis for comparison, PCB Aroclors were not detected in the reference surface soil samples collected from the CCSP upland. Total PCB concentrations were, however, calculable for one reference surface soil sample collected at CCSP (only using the NOAA method, as no Aroclors were detected). The total PCB concentration in this sample was significantly lower than the mean and maximum calculated total PCB concentrations for landfill area subsurface soils.

Aroclor and total PCB data for subsurface soil samples from the site are presented on Figures E-9 and E-10 in Appendix E. Bubble plots showing the concentrations of PCB Aroclors and total PCBs in subsurface soil samples collected from the site during the RI are presented as Figures F-235a through F-250c in Appendix F-11. Bubble plots were generated for both the 1 to 4 ft bgs interval and the >4 ft bgs interval. The description of the nature and extent of PCBs in landfill area subsurface soils above integrates the entire subsurface zone below 1 ft bgs. The box and whisker plots presented as Figures H-67a through H-74c in Appendix H-3 demonstrate the relationships between data for various Aroclors and total PCBs in landfill area subsurface soils and Aroclor and total PCB data for landfill area surface soils.

Pesticides

delta-HCH and toxaphene were not detected in any of the subsurface soil samples collected from the landfill portion of the site and evaluated for pesticides. The following pesticides were detected in 5% or fewer of all subsurface soil samples from the landfill area: aldrin; *beta*-BHC; endrin; endrin ketone; heptachlor; heptachlor epoxide; and methoxychlor.

alpha-BHC, endosulfan I, endosulfan sulfate, endrin aldehyde, and *gamma*-BHC (lindane) were detected in approximately 6% to 10% of the landfill area subsurface soil samples analyzed for pesticides. 2,4-DDE, 2,4-DDT, 4,4-DDT, *cis*-nonachlor, dieldrin, and endosulfan II were detected in approximately 20% to 50% of the subsurface soil samples from the landfill analyzed for pesticides. 2,4-DDD, *alpha*-chlordane, *gamma*-chlordane, and *trans*-nonachlor were detected in approximately 50% to 70% of the subsurface soil samples from the landfill that were analyzed for pesticides. 4,4-DDD and 4,4-DDE were detected in more than 90% of the subsurface soil samples collected from the landfill area of the site.

Detections of pesticides in subsurface soils appear widespread across the landfill portion of the site. In general, there does not appear to be a clear pattern in the occurrence of pesticides. However, in several cases, elevated levels of pesticides appear to correlate to areas of presumed historical waste disposal, including the former radioactive waste storage shed, drum disposal areas, and oil disposal areas. With the exception of dieldrin, for all pesticides detected in landfill area subsurface soils for which there exists a Region 9 PRG, the maximum detected concentration at the site was well below the relevant PRG. The maximum detected concentration of dieldrin was 48.3 µg/kg, which represents a relatively minor exceedance of the relevant PRG for this compound (30.4 µg/kg). The subsurface soil sampling location that exhibited this maximum dieldrin concentration is located in the vicinity of the former radioactive waste storage shed in the northwestern portion of the site.

For a limited number of pesticides, mean and maximum detected concentrations were relatively consistent between landfill area subsurface soils and surface soils. Also, the mean and maximum detected concentrations of heptachlor and methoxychlor were higher in landfill area surface soils compared to subsurface soils. However, in general, mean and maximum detected pesticide concentrations generally were appreciably higher in landfill area subsurface soils compared to landfill area surface soil samples.

Reference subsurface soil samples were not collected from the CCSP upland for the analysis of pesticides. However, surface soil samples were collected from the CCSP upland for analysis of pesticides, and provide a valuable basis for comparison. Several pesticides were detected in the CCSP surface soil samples. The number of pesticides detected in the CCSP upland surface soils was significantly less than the number of individual pesticides detected in subsurface soils from the landfill portion of the site. The maximum concentrations of those pesticides detected in both surface soils from the CCSP upland and subsurface soils from the landfill portion of the site were appreciably lower for the CCSP surface soils. There are no specific background values for pesticides in soils at Alameda Point.

Pesticide data for subsurface soil samples from the site are presented on Figures E-9 and E-10 in Appendix E. Bubble plots showing the concentrations of pesticides in subsurface soil samples collected from the site during the RI are presented as Figures F-251 through F-294 in Appendix F-12. Bubble plots were generated for both the 1 to 4 ft bgs interval and the interval below 4 ft bgs. The description of the nature and extent of pesticides in landfill area subsurface soils above integrates the entire subsurface beneath 1 ft bgs. The box and whisker plots presented as Figures H-75 through H-103 in Appendix H-4 demonstrate the relationships between data for various pesticides in landfill area subsurface soils and pesticide data associated with CCSP upland reference surface soils. These box and whisker plots also demonstrate the relationships between data for various pesticides in landfill area subsurface soils and landfill area surface soils.

VOCs

Several more VOCs were detected in the landfill area subsurface soil samples compared to landfill area surface soil samples. The following VOCs were detected in at least one subsurface soil sample collected from the landfill portion of the site: 1,1,2,2-tetrachloroethane; 1,2,3-trichlorobenzene; 1,2,4-trichlorobenzene; 1,2-dichlorobenzene; 1,3-dichlorobenzene; 1,4-dichlorobenzene; MEK; acetone; benzene; bromodichloromethane; carbon disulfide; chlorobenzene; *cis*-1,2-dichloroethene (1,2-DCE); cyclohexane; ethylbenzene; isopropyl benzene; xylenes; methylcyclohexane; methylene chloride; styrene; tetrachloroethene; toluene; and trichloroethene. Of these VOCs, 1,1,2,2-tetrachloroethane, 1,2,3-trichlorobenzene, 1,2,4-trichlorobenzene, 1,3-dichlorobenzene, bromodichloromethane, *cis*-1,2-DCE, cyclohexane, styrene, tetrachloroethene, toluene, and trichloroethene were detected in 5% or fewer of the subsurface soil samples. With the exception of toluene, the maximum detected concentrations of these VOCs were below 10 µg/kg. The maximum detected concentration of toluene was 73 µg/kg.

1,2-dichlorobenzene, MEK, benzene, carbon disulfide, ethylbenzene, xylenes, and methylcyclohexane were detected in 7 to 16% of the subsurface soil samples collected from the landfill that were analyzed for VOCs. The maximum detected concentrations of all of these compounds were below 50 µg/kg. 1,4-dichlorobenzene, chlorobenzene, and isopropyl benzene were detected in 40%, 39%, and 24% of the subsurface samples collected from the landfill portion of the site, respectively. The maximum detected concentrations of these compounds were 50 µg/kg (1,4-dichlorobenzene), 95 µg/kg (chlorobenzene), and 42 µg/kg (isopropyl benzene).

Acetone was detected in approximately 76% of the subsurface soil samples analyzed for VOCs, with a maximum detected concentration of 160 µg/kg. Methylene chloride was detected in roughly 43% of the subsurface soil samples analyzed for VOCs, with a maximum detected concentration of 40 µg/kg. Both acetone and methylene chloride are common laboratory contaminants.

For VOCs that were detected in both landfill area subsurface and surface soil samples (i.e., acetone, MEK, and methylene chloride), the mean and maximum detected concentrations generally were consistent.

Overall, the occurrence of VOCs in subsurface soils in the landfill area is relatively widespread. However, VOCs occur at only very low concentrations. There is no clear evidence of significant source areas of VOCs in soil or of localized regions of significantly elevated concentrations. The maximum concentrations of all VOCs detected in subsurface soils in the landfill portion of the site were below Region 9 PRGs.

Specific background values have not been developed for VOCs in soils at Alameda Point. In addition, reference subsurface soil samples were not collected from the CCSP upland for the analysis of VOCs. Surface soil samples were collected from the CCSP upland for analysis of VOCs, and, although the data would provide a valuable basis for comparison, VOCs were not detected in the reference surface soil samples collected from the CCSP upland.

VOC data for subsurface soil samples from the site are presented on Figures E-11 and E-12 in Appendix E. Bubble plots showing the concentrations of VOCs in subsurface soil samples collected from the site during the RI are presented as Figures F-295 through F-342 in Appendix F-16. Bubble plots were generated for both the 1 to 4 ft bgs interval and the interval below 4 ft bgs. The description of the nature and extent of VOCs in landfill area subsurface soils above integrates the entire subsurface zone below 1 ft bgs. The box and whisker plots presented as Figures H-104 through

H-127 in Appendix H-5 demonstrate the relationships between data for VOCs in landfill area subsurface soils and landfill area surface soils.

PCDD/PCDF

Several individual PCDD/PCDF congeners were detected in most of the subsurface soil samples collected from the landfill area of the site that were analyzed for these compounds. As with surface soil samples, the individual PCDD/PCDF congeners detected in discrete subsurface soil samples were summed (as described in Section 5.1) to generate a TEQ for each sampling point.

For the majority of subsurface soil sampling locations where PCDDs/PCDFs were analyzed in the landfill portion of the site, the calculated TEQ exceeds the Region 9 PRG for 2,3,7,8-TCDD. The highest calculated TEQs appear to be generally confined to the central portion of the landfill area. In some instances, the highly elevated TEQs in landfill area subsurface soils appear to be collocated with presumed areas of specific historical waste disposal (i.e., drum disposal areas). Overall, it appears that elevated levels of PCDDs/PCDFs are widespread in subsurface soils across the landfill portion of the site.

For all individual PCDDs/PCDFs and calculated TEQs, mean and maximum compound concentrations were either marginally or appreciably higher for landfill area subsurface soils compared to landfill area surface soils.

Reference subsurface soil samples were not collected from the upland portion of CCSP. In addition, reference surface soil samples were collected from the upland portion of CCSP, but were not analyzed for PCDD/PCDF. However, specific sampling was completed during the RI to evaluate the potential presence and extent of PCDD/PCDF contamination at Alameda Point reference locations (i.e., locations on Alameda Point but not affected in any known way by historical industrial or waste disposal activities). Specifically, soil sampling was completed at the same Alameda Point reference locations as were assessed during the radiological survey program (see Section 5.2.3 and TTFW, 2005). Although the reference sampling conducted at Alameda Point for PCDD/PCDF focused on surface soils only, a comparison between the reference results and the subsurface soil data from the landfill is nevertheless valuable. Several individual PCDD/PCDF congeners were detected in most of the surface soil samples collected from the Alameda Point reference locations. However, the concentrations of individual PCDD/PCDF congeners detected in the Alameda Point reference samples generally were significantly lower than those detected in subsurface soil samples collected from the landfill portion of the site. Similarly, TEQs calculated for the sampling locations in the Alameda Point reference areas were significantly lower than TEQs calculated for subsurface soil sampling locations in the landfill portion of the site. No calculated TEQs for reference area surface soil samples exceeded the Region 9 PRG for 2,3,7,8-TCDD.

TEQs calculated for subsurface soil samples from the site are presented on Figures E-9 and E-10 in Appendix E. Figures F-343 and F-344 in Appendix F-14 are bubble plots of TEQs generated for subsurface soil samples collected from the site during the RI. These bubble plots represent both the 1 to 4 ft bgs interval and the interval below 4 ft bgs. The description of the nature and extent of PCDD/PCDF in landfill area subsurface soils above integrates the entire subsurface zone below 1 ft bgs. The box and whisker plots presented as Figures H-128 through H-146 in Appendix H-6 demonstrate the relationships between data for various PCDDs/PCDFs and TEQs in landfill area subsurface soils and the reference PCDD/PCDF and TEQ data developed for Alameda Point. These box and whisker plots also demonstrate the relationships between data for various PCDDs/PCDFs and TEQs in landfill area subsurface soils and landfill area surface soils.

Petroleum Hydrocarbons

DRO and RRO were both detected in approximately 88% of the subsurface soil samples collected from the landfill area that were analyzed for these compounds. GRO were detected in roughly 67% of all subsurface soil samples collected from the landfill area that were analyzed for these compounds. The maximum concentrations of DRO, GRO, and RRO detected in landfill area subsurface soil samples were 21,000, 1,600, and 14,000 mg/kg, respectively. In general, detected concentrations of petroleum hydrocarbons appear to be appreciably higher in landfill area subsurface soil samples compared to landfill area surface soil samples.

Relatively elevated concentrations of petroleum hydrocarbons appear to be somewhat widespread in subsurface soils in the landfill area of the site. In addition, some of the most elevated levels of petroleum hydrocarbons in subsurface soils were detected in presumed areas of specific historical waste disposal, including former drum, oil, and pesticide disposal locations.

Specific background values have not been developed for TPHs in soils at Alameda Point, and PRGs do not exist for DRO, GRO, or RRO. Subsurface soil samples were not collected from the CCSP upland as part of the reference sampling program during the RI, and TPHs were not analyzed for in the reference surface soil samples collected from the CCSP upland.

TPH data for subsurface soil samples from the site are presented on Figures E-9 and E-10 in Appendix E. Bubble plots showing the concentrations of petroleum hydrocarbons in subsurface soil samples collected from the site during the RI are presented as Figures F-345 through F-350 in Appendix F-14. Bubble plots were generated for both the 1 to 4 ft bgs interval and the interval below 4 ft bgs. The description of the nature and extent of petroleum hydrocarbons in landfill area subsurface soils above integrates the entire subsurface zone below 1 ft bgs. The box and whisker plots presented as Figures H-147 through H-149 in Appendix H-7 demonstrate the relationships between data for petroleum hydrocarbons in landfill area subsurface soils and landfill area surface soils.

Radionuclides

Pb-210 was detected in only two of the 15 landfill area subsurface soil samples evaluated for this analyte, with a maximum concentration of 1.07 pCi/g. Gross alpha was detected in 14 of the 15 subsurface soil samples from the landfill that were analyzed for this analyte, at a maximum concentration of 18 pCi/g. Gross beta, U-234, and U-238 were detected in all of the landfill area susbsurface soil samples evaluated for these radionuclides, with maximum concentrations of 37 pCi/g, 0.49 pCi/G, and 0.53 pCi/g, respectively. Overall, gross alpha, gross beta, U-234, and U-238 were detected primarily in areas of presumed discrete historic waste disposal, including the vicinity of the former radioactive waste storage shed. However, as discussed in Section 4.0, samples for analysis of radionuclides were selected primarily from locations that corresponded to areas of suspected specific historical waste disposal.

Ra-228 was detected in all of the landfill area subsurface soil samples analyzed for this constituent, whereas Ra-226 was detected in approximately 65% of the subsurface soil samples analyzed for this compound. The maximum concentration of Ra-228 detected in subsurface soils from the landfill portion of the site was 0.57 pCi/g. The minimum concentration of Ra-226 detected in subsurface soils from the landfill area of the site was approximately 0.3 pCi/g (i.e., roughly equal to the background value developed during the radiological survey described in Section 5.2.3 and TTFW, 2005) and the maximum concentration detected was approximately 1.2 pCi/g. The highest Ra-226 concentrations detected in landfill area subsurface soil samples appear to be isolated to the eastern

central portion of the landfill, a small area in the southern portion of the landfill, and the vicinity of the former radioactive waste storage shed.

In general, the detected concentrations of radionuclides in landfill area subsurface soils are highly consistent with detected concentrations of these compounds in landfill surface soils. No specific background values have been developed for radionuclides in soils at Alameda Point other than the value generated by the Navy during its recent survey program (see Section 5.2.3). Reference subsurface soil samples were not collected from the CCSP upland, and radionuclides were not analyzed for in the reference surface soil samples collected from the CCSP upland. In addition, there are no PRGs for radionuclides.

Radionuclide data for subsurface soil samples from the site are presented on Figures E-5 and E-4 in Appendix E. Bubble plots showing the concentrations of radionuclides in subsurface soil samples collected from the site during the RI are presented as Figures F-351 through F-366 in Appendix F-15. Bubble plots were generated for both the 1 to 4 ft bgs interval and the interval below 4 ft bgs. The description of the nature and extent of radionuclides in landfill area subsurface soils above integrates the entire subsurface below 1 ft bgs. The box and whisker plots presented as Figures H-150 through H-157 in Appendix H-8 demonstrate the relationships between data for radionuclides in landfill area subsurface soils and landfill area surface soils.

Explosives

Only two explosive constituents were detected in subsurface soil samples collected from the landfill portion of the site that were analyzed for these compounds. Hexahydro-1,3,5-trinitro-1,3,5-triazine (RDX) was detected in one of the four subsurface soil samples collected from the landfill area that were analyzed for explosives, and octahydro-1,3,5,7-tetranitro-1,3,5,7-tetrazocine (HMX) was detected in two of the four subsurface soil samples collected from the landfill area that were analyzed for explosive constituents. The maximum concentrations of RDX and HMX detected in landfill area subsurface soils were 0.13 mg/kg and 0.38 mg/kg, respectively, which are well below the Region 9 PRGs for these compounds. The only location where the explosive constituents were detected was in the southeastern corner of the landfill. However, this is the only location in which subsurface soil samples were analyzed for explosive constituents.

HMX and RDX were not detected in landfill area surface soil samples. No specific background values have been developed for explosives in soils at Alameda Point. Reference subsurface soil samples were not collected from the CCSP upland, and explosives were not analyzed for in the reference surface soil samples collected from the CCSP upland.

Explosives data for soil samples from the site are presented on Figures E-11 and E-12 in Appendix E. Bubble plots showing the concentrations of explosives in subsurface soil samples collected from the site during the RI are presented as Figures F-376 through F-370 in Appendix F-16. Bubble plots were generated for both the 1 to 4 ft bgs interval and the interval below 4 ft bgs. The description of the nature and extent of explosives in landfill area subsurface soils above integrates the entire subsurface zone below 1 ft bgs. The box and whisker plots presented as Figures H-158 through H-161 in Appendix H-9 demonstrate the relationships between data for explosives in landfill area subsurface soils and landfill area surface soils.

General Chemistry

Several subsurface soil samples collected from the landfill portion of the site were evaluated for moisture content, TOC, grain-size distribution, and/or sulfides. These analytical parameters are, in

and of themselves, not indicative of any type or distribution of contamination in landfill area subsurface soils, and therefore are not discussed here.

5.3.1.4   Landfill Soil Gas

During the winter 2004 quarterly monitoring program at IR Site 2, a total of 13 soil gas samples were collected from individual monitoring probes installed in the landfill area at depths ranging from 0.5 ft to 9 ft bgs. All samples were analyzed for VOCs and gasoline range TPHs.

Table 5-6 provides a summary of the soil gas data generated from the fall 2002 through winter 2004, during the Navy's quarterly monitoring program at the site (ITSI, 2005). As discussed in Section 5.1, the soil gas data generated in winter 2004 are used in this RI Report to characterize the nature and extent of contamination in soil gas.

VOCs and TPH-G

Acetone and TPH-gasoline were detected in approximately 80% and 100%, respectively, of all soil gas samples collected from the landfill in winter 2004. Benzene, chlorobenzene, toluene and xylenes were detected in roughly 50% of all soil gas samples collected. Ethylbenzene was detected in approximately 31% of all soil gas samples collected. Naphthalene, chloroethane and *cis*-1,2-DCE were detected in 23% of all soil gas samples collected, and tetrachloroethene (PCE) and chloroform were detected in 15% of all soil gas samples collected. The following chemicals were not detected in any soil gas samples: 1,1,1-trichloroethane (1,1,1-TCA), 1,1,2,2,-tetrachloroethane, 1,1,2-TCA, 1,1-dichloroethane (1,1-DCA), 1,1-DCE, 1,2-DCA, 1,2-dichloropropane, 2-butanone, 2-hexanone, 4-methyl-2-pentanone, bromodichloromethane, bromoform, bromomethane, carbon tetrachloride, chloromethane, *cis*-1,3-dichloropropene, dibromochloromethane, methyl-*tert*-butyl ether (MTBE), methylene chloride, styrene, *trans*-1,2-DCE, *trans*-1,3-dichloropropene, trichloroethene (TCE), vinyl acetate, and vinyl chloride.

The majority of maximum concentrations of VOCs in soil gas were detected in the soil gas monitoring probes that are located in the deeper intervals of the landfill. One exception is the maximum concentration of naphthalene (4,500 ppbv) detected in the shallow interval of MG2-04, which is located in the northwestern portion of the landfill near a potential chemical drum disposal area. Analytical data collected in this area indicate that naphthalene is present at low levels in surface soil and groundwater (9.2 µg/kg in surface soil at sampling location SOC-24 and 6 µg/L in groundwater at sampling location HYP-10). The highest concentrations of benzene (460 ppbv) and chlorobenzene (900 ppbv) were found at medium and deep intervals of MG2-02, respectively, which is located in the south-central portion of the landfill near the potential pesticides disposal area. Benzene and chlorobenzene were either not detected or detected at low levels in soil and groundwater samples collected in this area.

Acetone, methylene chloride, naphthalene, and TPH-gasoline were the only chemicals detected (and at very low concentrations) in soil gas from MG2-05, which is located in the north-central portion of the landfill near a potential drum disposal area. At MG2-01 (i.e., the soil gas monitoring probe located closest to the boundary between the landfill and wetland), acetone and TPH-gasoline were detected at 8.7 ppbv and 0.072 ppmv, respectively, in the shallow interval (0.5 to 1.8 ft bgs). In the deep interval for the same probe (2.5 to 4 ft bgs), acetone and TPH-gasoline were detected at 61 ppbv and 0.2 ppmv, respectively.

There are no existing soil gas background data, and soil gas data were not collected for this RI. Although the existing soil gas data are useful in determining the conditions in soil gas and whether

there are any potential source areas (or hot spots) at the site, the significant amount of soil and groundwater data that were collected through the RI sampling activities are used to develop a comprehensive understanding of the nature and extent of contamination at the site. Soil and groundwater samples were collected close to the soil gas monitoring probes where VOCs were detected in winter 2004, and these soil and groundwater samples do not indicate there are significant VOC contamination sources at the site (see Sections 5.3.1.2 and 5.3.1.3). All available historical soil gas data (including that collected in winter 2004) is used to evaluate the potential for vapor intrusion to indoor air in the human health risk assessment (see Section 6.3.3).

A site map showing benzene, chlorobenzene, and naphthalene concentrations detected in soil gas samples collected from the site during the winter 2004 sampling event is presented as Figure E-13 in Appendix E.

5.3.1.5 Landfill FWBZ Groundwater

During the course of the RI at IR Site 2, a total of 14 FWBZ groundwater samples were collected from the landfill area of the site. Samples were generated during the dry and wet season sampling programs, and were collected from the shallow portion of the FWBZ using temporary wellpoints. In addition, recent quarterly groundwater monitoring completed through the Navy's ongoing monitoring program (winter 2004) coincides very closely in time with the dry season sampling event of the RI (ITSI, 2005). Ground-water samples collected through the ongoing monitoring program come from both FWBZ and SWBZ wells located along the periphery of the site and the berm between the landfill and wetlands. The landfill area FWBZ groundwater data generated specifically through the RI and data from the recent quarterly sampling event were combined to develop the evaluation of the nature and extent of groundwater contamination in the FWBZ at the site. The following analytical parameters were evaluated in the FWBZ groundwater samples collected from the landfill, combining the RI specific data and the quarterly monitoring data described above (as noted within specific compound classes, individual analytes evaluated during the RI and the quarterly monitoring program varied to some degree):

- 27 groundwater samples were analyzed for VOCs (note that 14 groundwater samples were analyzed for 1,1,2-trichloro-1,2,2-trifluoroethane, *cis*-1,3-dichloropropene, cyclohexane, methyl acetate, methylcyclohexane, and *trans*-1,3-dichloropropene; and 13 groundwater samples were analyzed for tert-amyl methyl ether, TBA, and tert-butyl ethyl ether);

- 26 groundwater samples were analyzed for metals (note that 12 groundwater samples were analyzed for sodium), pesticides (note that 14 groundwater samples were analyzed for 2,4-DDD, 2,4-DDE, 2,4-DDT, *cis*-nonachlor, and *trans*-nonachlor, and 12 groundwater samples were analyzed for 2,4,5,6-tetrachloro-*m*-xylene), and PCBs;

- 25 groundwater samples were analyzed for SVOCs/PAHs (note that 14 groundwater samples were analyzed for 1,4-dioxane, 13 groundwater samples were analyzed for 1,1-biphenyl, 1,2,4,5-tetrachlorobenzene, acetophenone, atrazine, benzaldehyde, caprolactam, and carbazole, and 12 groundwater samples were analyzed for 1-methylnaphthalene, benzoic acid, and *n*-nitrosodimethylamine [NDMA]);

- 21 groundwater samples were analyzed for TPHs and alkalinity;

- 10 groundwater samples were analyzed for gross alpha, gross beta, and Ra-228;

- 7 groundwater samples were analyzed for hexavalent chromium, chloride, nitrate, nitrite, sulfate, and sulfide;

- 6 groundwater samples were analyzed for tritium;

- 5 groundwater samples were analyzed for Pb-210, Ra-226, U-234, U-235, and U-238;

- 3 groundwater samples were analyzed for explosive constituents (note that 15 groundwater samples were analyzed for 2,4-dinitrotoluene, 2,6-dinitrotoluene, and nitrobenzene); and,

- 2 groundwater samples were analyzed for PCDDs/PCDFs.

Table 5-7 provides a general statistical summary of the groundwater data used to interpret the nature and extent of contamination in the FWBZ in the landfill portion of the site. During the RI, landfill area FWBZ groundwater samples were analyzed for total and dissolved fractions of all compounds other than VOCs, PCDDs/PCDFs, and general chemistry parameters. The quarterly groundwater data for landfill area FWBZ wells are only available for the total fraction for all analytes.

Metals

All metals analyzed for were detected in at least one groundwater sample from the FWBZ in the landfill area. Aluminum, arsenic, barium, copper, iron, lead, magnesium, manganese, nickel, and zinc were detected in all of the FWBZ groundwater samples collected from the landfill area. Antimony, beryllium, cadmium, chromium, cobalt, molybdenum, silver, and vanadium were detected in 46% to 65% of the landfill area FWBZ groundwater samples. Thallium, selenium, and mercury were detected in 38%, 27%, and 19% of the landfill area FWBZ groundwater samples, respectively. Sodium was detected in all of the groundwater samples from the landfill area FWBZ that were analyzed for this metal, and hexavalent chromium was detected in 86% of the samples analyzed for this metal (i.e., 6 of the 7 samples).

Maximum detected concentrations of all metals other than beryllium, hexavalent chromium, silver, sodium, and thallium exceeded established background values for metals in Alameda Point groundwater. Exceedances of antimony, arsenic, cadmium, cobalt, mercury, molybdenum, and vanadium were relatively minor in comparison to the established background values for these compounds. Maximum detected concentrations of antimony, arsenic, iron, and manganese exceeded Region 9 PRGs (note that, as with soils, the background concentration developed for arsenic in Alameda Point groundwater is higher than the relevant PRG). In addition, arsenic, copper, lead, nickel, and zinc concentrations were detected above chronic marine AWQC values, and copper and zinc exceeded acute marine AWQC values. Overall, concentrations of metals in filtered groundwater samples collected from landfill FWBZ locations were significantly lower compared to unfiltered samples, indicating that suspended solids entrained in the samples collected from the temporary wellpoints during the RI contributed to detected metals concentrations in the total fraction.

Higher concentrations of arsenic generally were found in the wells along the southwestern and southeastern border of the landfill. Cadmium, copper, lead, manganese, mercury, nickel, and zinc generally exhibited higher concentration in the wells located in the northern portion of the landfill. Chromium and magnesium had higher concentrations in both the northwest and southwest portions of the landfill and along the landfill/wetland border. Detected concentrations of metals in landfill-area FWBZ groundwater coincided with areas that exhibited detectable concentrations of these metals in surface and/or subsurface soils. As discussed above, lead is characterized by two areas in landfill

subsurface soil exhibiting highly elevated concentrations. However, concentrations of lead in landfill-area FWBZ groundwater are generally not excessively elevated (maximum 200 µg/L in the total fraction and only approximately 9 µg/L in the dissolved fraction), and the maximum detected concentration of lead (both total and dissolved) in FWBZ groundwater does not coincide with either of the two areas demonstrating highly elevated lead concentrations in subsurface soil. Overall, given that detectable concentrations of metals in both landfill soil and FWBZ groundwater are widespread, that concentrations detected in the landfill FWBZ generally were not significantly elevated, and that dissolved-phase metals concentrations were significantly lower than in the total (i.e., unfiltered) fraction, there do not appear to be any distinct hotspots or obvious source areas for metals in the FWBZ in the landfill area.

Metals data for FWBZ groundwater samples from the site are presented on Figure E-14 in Appendix E. Bubble plots showing the concentrations of metals in FWBZ groundwater samples collected from the site are presented as Figures F-371 through F-412 in Appendix F-17. Bubble plots were generated for the total and dissolved fraction of each detected analyte. The box and whisker plots presented as Figures H-162 through H-184 in Appendix H-10 demonstrate the relationships between data for various metals in landfill and wetland area FWBZ groundwater. These box and whisker plots also demonstrate the relationship between total and dissolved metals data.

SVOCs/PAHs

The only SVOC/PAH detected in 100% of the FWBZ groundwater samples from the landfill area was 1-methylnaphthalene. 1,4-dioxane, anthracene, benzaldehyde, fluoranthene, fluorene, naphthalene, phenanthrene, and pyrene were detected in between 68% and 96% of the landfill area FWBZ groundwater samples. 1,1-biphenyl, 2-chlorophenol, 2-methylnaphthalene, acenaphthylene, benzo(a)anthracene, benzoic acid, *bis*(2-ethylhexyl) phthalate, carbazole, chrysene, dibenzofuran, and *n*-nitrosodiphenylamine were detected in between 36% and 67% of the groundwater samples. 4-methylphenol, benzo(a)pyrene, benzo(b)fluoranthene, benzo(g,h,i)perylene, and indeno(1,2,3-cd)pyrene were detected in 15% to 35% of groundwater samples from the landfill area FWBZ. 2,4-dichlorophenol, 2,4-dimethylphenol, 2-methylphenol, benzo(k)fluoranthene, butyl benzyl phthalate, caprolactam, isophorone, and phenol had at least one detection but were detected in less than 15% of the groundwater samples. All other SVOCs/PAHs were not detected in groundwater samples from the landfill area FWBZ.

Of the 37 SVOCs/PAHs detected in the landfill area FWBZ, only benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, *bis*(2-ethylhexyl) phthalate, carbazole, dibenzofuran, indeno(1,2,3-cd) pyrene, naphthalene, and *n*-nitrosodiphenylamine exhibited maximum concentrations exceeding Region 9 PRGs. The maximum concentration of benzo(a)pyrene (0.13 µg/L) and naphthalene (140 µg/L) exceeded the Region 9 PRGs (0.01 µg/L and 6.2 µg/L, respectively) by more than an order of magnitude. Maximum concentrations of other SVOCs/PAHs exceeded their respective PRGs by 1.1 to 4.5 times.

The maximum detected concentrations were greater for the total fraction than the dissolved fraction for all detected SVOC/PAHs with the exception of the following compounds: 1,4-dioxane, 2,4,6-trichlorophenol, 2-methylphenol, acetophenone, anthracene, diethyl phthalate, isophorone, and *n*-nitrosodiphenylamine. The majority of the SVOCs/PAHs detected in landfill area FWBZ demonstrated consistent concentrations in both the total and dissolved fractions. When considering the dissolved fraction, only carbazole, dibenzofuran, naphthalene, and *n*-nitrosodiphenylamine exceeded their respective PRGs (by 2.9, 3.9, 19.4, and 1.02 times, respectively). Some SVOCs/PAHs, including 2,4,6-trichlorophenol, acetophenone, and diethyl phthalate, were present in the dissolved fraction of the groundwater but not the total fraction.

In general, SVOCs/PAHs appear to be relatively widespread throughout FWBZ groundwater in the landfill. However, one sampling location in the south central portion of the landfill (HYP11) showed the greatest number of individual SVOCs/PAHs detected at higher concentrations, including six analytes at concentrations exceeding Region 9 PRGs. In fact, the maximum detected concentrations of 13 of the 37 SVOCs/PAHs identified in the landfill area FWBZ were found in HYP11. This sampling location was in the potential asbestos disposal area. Soil sampling data from a location corresponding directly to HYP11 demonstrate moderate concentrations of SVOCs/PAHs that are well below the maximum detected SVOC/PAH concentrations for landfill area soil. Overall, given that detectable concentrations of SVOCs/PAHs in both landfill soil and FWBZ groundwater are widespread, maximum concentrations detected in the landfill FWBZ do not correspond to areas where maximum soil concentrations were observed, and dissolved phase SVOC/PAH concentrations were significantly lower than in the total (i.e., unfiltered) fraction, there do not appear to be any distinct hot spots or obvious source areas for SVOCs/PAHs in the FWBZ in the landfill area.

SVOC/PAH data for FWBZ groundwater samples from the site are presented on Figure E-15 in Appendix E. Bubble plots showing the concentrations of SVOCs/PAHs in groundwater samples collected from the site are presented as Figures F-413 through F-475 in Appendix F-18. Bubble plots were generated for the total and dissolved fraction of each detected analyte. The box and whisker plots presented as Figures H-185 through H-225 in Appendix H-11 demonstrate the relationships between data for various SVOCs/PAHs in landfill area groundwater. These box and whisker plots also demonstrate the relationship between total and dissolved SVOC/PAH data. AWQC and Alameda Point background values are not available for SVOCs/PAHs in groundwater.

PCBs

Two Aroclors were detected in FWBZ groundwater in the landfill area of the site, and only 2 of 26 sampling locations evaluated for PCBs exhibited Aroclor detections. Location HYP01 exhibited a detection of Aroclor 1260 at approximately 0.3 μg/L (average of duplicate samples), and HYP02 exhibited a detection of Aroclor 1232 at approximately 11.7 μg/L. These detections occurred only in the total fraction and not in the dissolved fraction of the samples, consistent with the fact that PCBs tend to be strongly sorbed to solid matrix material and not readily dissolved. Overall, PCBs were detected in only a small number of locations, and the detected concentrations of PCBs were quite low in the landfill area FWBZ. There are no Region 9 PRGs for the individual Aroclors that were detected in landfill area FWBZ groundwater samples. However, the detected concentrations of Aroclors 1232 and 1260 exceeded the highly conservative chronic marine AWQC for these individual compounds.

Calculating total PCBs as the sum of Aroclors was only possible at locations HYP01 and HYP02 (i.e., the only locations where individual Aroclors were detected in groundwater), and the maximum detected total PCB concentration was obviously identical to the approximately 11.7 μg/L of Aroclor 1232 detected at HYP02. The concentrations of total PCBs calculated as the sum of Aroclors for the landfill area FWBZ both exceeded the highly conservative Region 9 PRG and the chronic marine AWQC. Furthermore, sampling location HYP02 specifically targeted a presumed historical drum disposal area.

Using the NOAA method, total PCBs were actually calculable for 79% of the landfill area FWBZ groundwater samples (i.e., 11 out of 14 samples for which a PCB congener dataset existed) for the total fraction and for 71% of the groundwater samples (i.e., 10 of 14 samples) for the dissolved fraction. Calculable concentrations of total PCBs generally were confined to the central and northern portions of the landfill, as well as the interior margin (including the area of the former radioactive waste storage shack). The maximum total PCB concentration based on the NOAA method was

approximately 1.9 µg/L for the total fraction and 0.8 µg/L for the dissolved fraction, also consistent with the strong sorption characteristics of PCBs. All of the calculated concentrations of total PCBs (total and dissolved) for the landfill area FWBZ were quite low, but exceeded the highly conservative Region 9 PRG and the chronic marine AWQC. Exceedances of these conservative benchmarks did not occur in FWBZ groundwater in the southern portion of the landfill.

At least some of the NOAA-based concentrations of total PCBs in FWBZ groundwater appear to coincide with potential discrete historical disposal areas. For example, groundwater from the FWBZ in the oil pit in the northeastern portion of the site (i.e., HYP03) exhibited a concentration of approximately 1.9 µg/L, and FWBZ groundwater in the chemical drum area to the north of the oil pit (i.e., HYP02) exhibited a concentration of approximately 1.2 µg/L. It should be noted that the majority of the landfill area FWBZ groundwater samples were collected intentionally in or near the potential waste disposal areas. In addition, groundwater collected from outside the potential disposal areas did not necessarily exhibit the lowest concentrations of PCBs. In fact, three locations with no detected PCBs were located in potential disposal areas in the southern part of the landfill.

At location HYP12, total PCBs were not calculable in the total fraction, whereas total PCBs were calculated for the dissolved fraction and exceeded the Region 9 PRG and chronic marine AWQC. Conversely, at location HYP01 the calculated total PCB concentration in the total fraction exceeded the Region 9 PRG and chronic marine AWQC but total PCBs were not calculable for the dissolved fraction. For the majority of landfill area FWBZ groundwater sampling locations, dissolved total PCB concentrations were significantly lower than the total concentrations.

The two locations where individual PCB Aroclors were detected (i.e., HYP01 and HYP02) correspond to locations where PCBs were detected in landfill area soils. In fact, a relatively substantial concentration of PCBs was detected at soil sampling loction SOC02, which corresponds to groundwater sampling location HYP01. However, given that detectable concentrations of PCBs in landfill FWBZ groundwater are very limited, concentrations detected in the landfill FWBZ were generally not significantly elevated, and PCB Aroclor concentrations were only identified in the total (i.e., unfiltered) fraction, it does not appear that landfill soils act in any way as a source for PCBs in the FWBZ in the landfill area.

PCB data for FWBZ groundwater samples from the site are presented on Figure E-16 in Appendix E. Bubble plots showing the concentrations of PCBs in groundwater samples collected from the site are presented as Figures F-476 through F-480b in Appendix F-19. Bubble plots were generated for the total and dissolved fraction. The box and whisker plots presented as Figures H-226 through H-229b in Appendix H-12 demonstrate the relationships between data for various PCBs and total PCBs in landfill area groundwater. These box and whisker plots also demonstrate the relationship between total and dissolved PCB data. Specific Alameda Point background values have not been developed for PCBs in groundwater.

Pesticides

All pesticides analyzed were detected in at least one landfill area FWBZ groundwater sample with the exception of 2,4-DDE, 2,4-DDT, endosulfan I, and toxaphene. 2,4,5,6-tetrachloro-*m*-xylene was detected in all samples collected during the quarterly monitoring event. The following pesticides were detected in between 15% and 27% of the samples: 2,4-DDD; 4,4-DDD; 4,4-DDE; aldrin; endosulfan II; *gamma*-chlordane; heptachlor; and heptachlor epoxide. The remaining pesticides, including 4,4-DDT, *alpha*-chlordane, *alpha*-BHC, *beta*-BHC, *cis*-nonachlor, *delta*-HCH, dieldrin, endosulfan sulfate, endrin, endrin aldehyde, endrin ketone, *gamma*-BHC, methoxychlor, and *trans*-nonachlor were detected in less than 15% of the samples for the total fraction.

Nine individual pesticides were detected at concentrations exceeding Region 9 PRGs or marine AWQC. These pesticides are 4,4-DDT, *alpha*-chlordane, *alpha*-BHC, aldrin, *beta*-BHC, dieldrin, endrin, heptachlor, and heptachlor epoxide. The chronic marine AWQC was exceeded by more than one order of magnitude for 4,4-DDT, dieldrin, endrin, and heptachlor. Concentrations of dieldrin and heptachlor also significantly exceeded Region 9 PRGs. Other pesticides exceeded AWQC and PRGs by less significant margins.

Only 2,4-DDD, 4,4-DDD, and aldrin were detected in at least one sample in the dissolved ground-water fraction. There is no PRG or AWQC value for 2,4-DDD, and 4,4-DDD was detected at a maximum dissolved concentration well below its PRG. Concentrations of aldrin exceeded the Region 9 PRG but do not have an applicable AWQC.

For at least a few pesticides in the total groundwater fraction, the distribution of concentrations can be mapped using isocontours that suggest plume behavior. Specifically, heptachlor and heptachlor epoxide demonstrate this kind of behavior, with the total fraction plumes generally located along the landfill/wetland boundary and in the northwestern corner of the site. However, the plumes do not correlate directly to locations where these compounds were detected at their highest levels in soil, and given that pesticide concentrations were only identified in the total (i.e., unfiltered) fraction except for low concentrations of two individual compounds, it does not appear that landfill soils act as a source for pesticides in the FWBZ in the landfill area. Rather, the "plumes" of pesticides appear to be related to the presence of these compounds in suspended solids entrained in turbid temporary well samples. This is consistent with the strong matrix-sorbing characteristics typical of pesticides.

Pesticide data for FWBZ groundwater samples from the site are presented on Figure E-16 in Appendix E. Bubble plots showing the concentrations of pesticides in groundwater samples collected from the site are presented as Figures F-481 through F-506 in Appendix F-20. Bubble plots were generated for the total and dissolved fraction of each detected analyte. The box and whisker plots presented as Figures H-230 through H-257 in Appendix H-13 demonstrate the relationships between data for various pesticides in landfill area groundwater. These box and whisker plots also demonstrate the relationship between total and dissolved pesticide data. In addition, isoconcentration maps for heptachlor and heptachlor epoxide in the total fraction of FWBZ groundwater are provided in Figures 5-2 and 5-3, respectively. Specific Alameda Point background values have not been developed for pesticides in groundwater.

VOCs

Twenty-seven VOCs were detected in at least one sample in FWBZ groundwater in the landfill area. The following VOCs were detected in 50% to 67% of the samples: 1,2-dichlorobenzene; 1,4-dichlorobenzene; acetone; benzene; chlorobenzene; cyclohexane; isopropylbenzene; methyl-cyclohexane; *ortho*-xylene; and toluene. 2-butanone, *cis*-1,2-DCE, ethylbenzene, isopropyl ether, *m,p*-xylenes, and TBA were detected in 15% to 41% of samples. VOCs found in less than 11% of the samples for the total fraction included: 1,1-DCA; 1,1-DCE; 1,2-DCA; carbon disulfide; chloroethane; styrene; *trans*-1,2-DCE; trichloroethene; trichlorofluoromethane; and vinyl chloride.

Region 9 PRGs were exceeded for the following VOCs: 1,4-dichlorobenzene; benzene; chloro-benzene; and vinyl chloride. Mean and maximum concentrations of 1,4-dichlorobenzene and benzene exceeded their respective PRGs by more than one order of magnitude, and vinyl chloride exceeded the relevant PRG by more than three orders of magnitude. However, vinyl chloride was detected only in one landfill area FWBZ groundwater sample. In addition, the PRG for all three of these VOCs is below 1 µg/L. The maximum detected concentration of chlorobenzene exceeded the applicable PRG by approximately 2.5 times.

The greatest number of individual VOCs were detected at HYP13 (located in the southeastern corner of the landfill) including the maximum detected concentrations of ethylbenzene, toluene, trichlorofluoromethane, and xylenes. However, this location does not appear to constitute a hot spot as only two analytes (benzene and 1,4-dichlorobenzene) exceeded relevant PRGs, and the exceedances of these two VOCs were widespread throughout the landfill area of the site. It appears that at least a few VOCs in landfill-area FWBZ groundwater display plume behavior. Specifically, benzene, chlorobenzene, and 1,4-dichlorobenzene appear to demonstrate this type of behavior, with the plumes generally occurring in the vicinity of the landfill/wetland boundary, the northwest corner of the site, and/or the north central portion of the landfill. Although VOCs were found to be generally nonexistent or related primarily to common laboratory contaminants in landfill surface soils, several VOCs were detected at relatively low levels in landfill area subsurface soils, including benzene, chlorobenzene, and 1,4-dichlorobenzene. Furthermore, the locations of the FWBZ groundwater plumes of these compounds are generally consistent with their distribution in landfill soil and the geometry of the plumes is generally consistent with groundwater flow at the site (see Section 2.0 of this RI Report).

VOC data for FWBZ groundwater samples from the site are presented on Figure E-17 in Appendix E. Bubble plots showing the concentrations of VOCs in groundwater samples collected from the site are presented as Figures F-507 through F-534 in Appendix F-21. The box and whisker plots presented as Figures H-258 through H-286 in Appendix H-14 demonstrate the relationships between data for various VOCs in landfill area FWBZ groundwater. In addition, isoconcentration maps for benzene, chlorobenzene, and 1,4-dichlorobenzene detected in landfill area FWBZ groundwater are provided in Figures 5-4, 5-5, and 5-6, respectively. AWQC and Alameda Point background values are not available for VOCs in groundwater.

PCDD/PCDF

PCDD/PCDF was analyzed in FWBZ groundwater from two temporary well locations in the landfill area during the March 2005 RI sampling activities (i.e., HYP22 in the western portion of the landfill and HYP23 in the eastern portion of the landfill). These sampling locations were completed to specifically target locations where the potential presence of PCDDs/PCDFs in landfill soil had previously been assessed. The samples were not filtered, so suspended solids associated with an undeveloped, temporary well point potentially affected the analytical results and overall representativeness of the samples. Nevertheless, 5 of 17 individual PCDD/PCDF congeners were detected in the samples and only two congeners, 1,2,3,4,6,7,8-heptachlorodibenzodioxin (HpCDD) and octachlorodibenzodioxin (OCDD), were detected in both. As with soil samples, the individual PCDD/PCDF congeners detected in groundwater samples were summed (as described in Section 5.1) to generate a TEQ for each sampling point. The TEQs calculated for the FWBZ groundwater results from the landfill were 0.5 and 11.7 picograms per liter (pg/L), both of which exceed the Region 9 tap water PRG for 2,3,7,8-TCDD of 0.45 pg/L. Reference TEQ groundwater data were not collected at CCSP or Alameda Point.

Given the strong matrix-sorbing characteristics typical of PCDDs/PCDFs, it is highly unlikely that landfill soils represent a source of these compounds to FWBZ groundwater. TEQs calculated for groundwater samples from the site are presented on Figures E-16 in Appendix E. Figure F-535 in Appendix F-22 is a bubble plot of TEQs generated the two FWBZ groundwater samples collected from the landfill.

Petroleum Hydrocarbons

DRO, GRO, and RRO were detected in 86%, 57%, and 86%, respectively, of the FWBZ groundwater samples collected from the landfill area of IR Site 2. The maximum concentrations of DRO, GRO, and RRO detected in landfill area FWBZ groundwater samples were 8,900, 2,700, and 8,800 µg/L, respectively. There are no specific Alameda Point background values, Region 9 PRGs, or AWQC values for TPHs in groundwater.

The highest concentrations of DRO were observed in the southern portion of the landfill along the border of the landfill and wetland. The maximum GRO concentration was found in the northeastern portion of the site in a potential chemical drum disposal area. The maximum RRO concentration is found in the southern portion of the site. In general, the more elevated concentrations of petroleum hydrocarbons in FWBZ groundwater do correlate to locations where appreciable detections of these compounds were identified in soils. However, although detections of petroleum hydrocarbons in soil were widespread across the landfill, the extent of elevated petroleum hydrocarbon concentrations in landfill FWBZ groundwater is not as extensive.

Dissolved concentrations of TPHs generally were significantly less than total concentrations. However, this comparison is complicated by the fact that quarterly groundwater sampling only includes analysis for total TPHs, and in several cases the maximum TPH concentration occurred in the quarterly monitoring dataset. In general, the total and dissolved concentrations of TPHs found in the temporary monitoring well locations were very similar.

TPH data for FWBZ groundwater samples from the site are presented on Figure E-16 in Appendix E. Bubble plots showing the concentrations of petroleum hydrocarbons in groundwater samples collected from the site are presented as Figures F-536 through F-541 in Appendix F-22. Bubble plots were generated for the total and dissolved fraction of each detected analyte. The box and whisker plots presented as Figures H-287 through H-289 in Appendix H-15 demonstrate the relationships between total and dissolved TPH data.

Radionuclides

Gross beta is the only radionuclide that was detected in every landfill area FWBZ groundwater sample. Ra-226, U-234, and U-238 were detected in between 60% and 80% of the groundwater samples from the landfill area FWBZ. Pb-210, Ra-228, and tritium were found in 20% or less of the groundwater samples. Gross alpha and U-235 were not detected in any FWBZ groundwater samples in the landfill area of the site.

The maximum concentrations of the radionuclides detected in landfill area FWBZ groundwater were: gross beta (174 pCi/L); Pb-210 (4.4 pCi/L); Ra-226 (0.91 pCi/L); Ra-228 (1.63 pCi/L); U-234 (1.41 pCi/L); U-238 (0.58 pCi/L); and tritium (355 pCi/L). There are no specific background values, Region 9 PRGs, or AWQC values for radionuclides in groundwater. In general, concentrations in the total and dissolved fractions were highly consistent for all detected radionuclides.

Radionuclide detections in landfill area FWBZ groundwater appear to be confined generally to the west central, northwest, and southern portions of the landfill area. Radionuclides also were detected in landfill soils in these areas. However, the concentrations of radionuclides detected in landfill area FWBZ generally were very low, and there do not appear to be any hot spots of radionuclides in the landfill area FWBZ.

Radionuclide data for FWBZ groundwater samples from the site are presented on Figure E-14 in Appendix E. Bubble plots showing the concentrations of radionuclides in groundwater samples collected from the site are presented as Figures F-542 through F-553 in Appendix F-23. Bubble plots were generated for the total and dissolved fraction of each detected analyte. The box and whisker plots presented as Figures H-290 through H-297 in Appendix H-16 demonstrate the relationships between total and dissolved radionuclide data.

Explosives

The only explosives detected in the FWBZ groundwater samples from the landfill portion of the site were 2,4-dinitrotoluene and 4-nitrotoluene. These analytes were both detected in only one groundwater sample (HYP12). 2,4-dinitrotoluene was not detected in the dissolved fraction, whereas 4-nitrotoluene was detected at a very similar concentration in both the total and dissolved fractions. The detected concentrations of both of these analytes were very low. However, the detected concentration of 4-nitrotoluene exceeded the Region 9 PRG by a very small margin for both the total and dissolved fraction.

Concentrations of explosive constituents were detected in landfill soil in the vicinity of HYP12, including 2,4-dinitrotoluene. However, several explosive constituents that were detected in landfill soil in this area were not detected in FWBZ groundwater. This is the only portion of the site that was assessed for explosives given the potential disposal of OEW material here, and the data generated do not suggest that there is a clear or significant source of explosives in groundwater.

Explosives data for FWBZ groundwater samples from the site are presented on Figure E-17 in Appendix E. Bubble plots showing the concentrations of radionuclides in groundwater samples collected from the site are presented as Figures F-554 and F-555 in Appendix F-24. Bubble plots were generated for the total and dissolved fraction of each detected analyte. The box and whisker plots presented as Figures H-298 through H-299 in Appendix H-17 demonstrate the relationships between total and dissolved explosives data.

General Chemistry

Several FWBZ groundwater samples in the landfill area were analyzed for alkalinity, chloride, nitrate, nitrite, sulfate, and sulfide. Chloride was detected in 100% of the samples, nitrate was detected in 29% of the samples, and sulfate was detected in 86% of the samples. The maximum concentration for chloride, nitrate, and sulfate were 12,000, 0.2, and 1,800 mg/L, respectively.

Chloride was primarily detected along the landfill/wetland boundary. The maximum sulfate concentration is found in the southern portion of the landfill. Nitrate was detected in one well in the far northern portion of the site. These analytical parameters are, in and of themselves, not indicative of any type or distribution of contamination in landfill area FWBZ groundwater, and therefore are not discussed in detail here.

5.3.1.6   Landfill SWBZ Groundwater

Groundwater in the SWBZ was evaluated using data exclusively from the recent quarterly monitoring dataset (i.e., winter 2004). A total of 10 groundwater samples were collected from wells in the landfill area that monitor the SWBZ. The following analytical parameters were evaluated in the SWBZ

groundwater samples collected from the landfill during winter 2004 based on the quarterly monitoring data:

- All 10 groundwater samples were analyzed for metals (including sodium), VOCs (note that no SWBZ samples were analyzed for 1,1,2-trichloro-1,2,2-trifluoroethane, *cis*-1,3-dichloropropene, cyclohexane, methyl acetate, methylcyclohexane, or *trans*-1,3-dichloropropene), SVOCs/PAHs (note that no SWBZ samples were analyzed for 1,1-biphenyl, 1,2,4,5-tetrachlorobenzene, 1,4-dioxane, 1-methylnaphthalene, acetophenone, atrazine, benzaldehyde, benzoic acid, caprolactam, carbazole, or NDMA), and TPHs; and,

- 7 groundwater samples were analyzed for gross alpha, gross beta, Ra-228, tritium, alkalinity, chloride, nitrate, nitrite, sulfate, and sulfide.

Table 5-7 provides a general statistical summary of the groundwater data used to interpret the nature and extent of contamination in the SWBZ in the landfill portion of the site. The landfill area SWBZ groundwater data are available only for the total fraction for all analytes.

Metals

Aluminum, barium, chromium, cobalt, copper, iron, lead, magnesium, manganese, nickel, sodium, vanadium, and zinc were detected in all of the SWBZ groundwater samples collected from the landfill area. Antimony, arsenic, and molybdenum were detected in 80% or more of the groundwater samples. Cadmium, selenium, and thallium were detected in roughly 10%, 50%, and 10% of the groundwater samples collected, respectively. Beryllium, mercury, and silver were not detected in any SWBZ groundwater samples from the landfill area.

Maximum concentrations of copper and nickel in the landfill SWBZ groundwater exceeded the chronic marine AWQC, but not the acute marine AWQC. Region 9 PRGs were exceeded for arsenic, iron, magnesium, and manganese. Alameda Point background data for metals in the SWBZ do not exist. The existing Alameda Point background dataset is specific to the FWBZ, and is therefore not entirely suitable for comparison to SWBZ metals data from the landfill.

Higher metals concentrations generally were observed along the southwestern border and south-eastern border with the landfill. Barium was an exception, with higher concentrations identified along the northwestern border between the landfill and wetland. The maximum concentration of iron was observed in the northwestern portion of the landfill. Overall, mean and maximum detected concentrations of metals were either consistent between landfill area SWBZ groundwater and landfill area FWBZ groundwater or appreciably lower for landfill area SWBZ groundwater (see Table 5-7). However, mean and maximum concentrations of magnesium and manganese were appreciably higher for the landfill area SWBZ.

Some locations where more elevated concentrations of metals were identified in landfill-area SWBZ groundwater correlate to locations where metals concentrations were detected in landfill-area FWBZ groundwater and/or soils. However, given that metals concentrations in the SWBZ were generally significantly lower than in the FWBZ, the relatively limited number of individual compounds that exceed relevant benchmarks in the SWBZ, and the overall limited magnitude of metals impact in landfill-area SWBZ groundwater, the distribution of metals in landfill area SWBZ groundwater does not suggest the presence of any discrete contamination hot spots or sources.

Metals data for SWBZ groundwater from the site are presented on Figure E-14 in Appendix E. Bubble plots showing the concentrations of metals in SWBZ groundwater samples are presented as Figures F-556 through F-574 in Appendix F-25. The box and whisker plots presented as Figures H-162 through H-184 in Appendix H-10 demonstrate the relationships between data for various metals in landfill area SWBZ and FWBZ groundwater. These box and whisker plots also demonstrate the relationships between total and dissolved metals data.

SVOCs/PAHs

Only two SVOCs were detected in SWBZ groundwater in the landfill area. 1-methylnaphthalene was detected in 100% of the samples and *bis*(2-ethylhexyl) phthalate was detected in 20% of the samples.

The mean and maximum concentration for 1-methylnaphthalene were 89 and 105 µg/L, respectively, and for *bis*(2-ethylhexyl) phthalate were both 1.3 µg/L. Detected *bis*(2-ethylhexyl) phthalate concentrations were below the Region 9 PRG, and there is no PRG for 1-methylnaphthalene. The mean and maximum detected concentrations of both of these analytes in the landfill-area SWBZ were highly consistent with those in the landfill-area FWBZ. These data do not suggest that SVOCs/PAHs represent a contamination issue in the landfill-area SWBZ.

SVOC/PAH data for SWBZ groundwater samples from the site are presented on Figure E-15 in Appendix E. Bubble plots showing the concentrations of SVOCs/PAHs in SWBZ groundwater samples collected from the site are presented as Figures F-575 and F-576 in Appendix F-26. The box and whisker plots presented as Figures H-185 through H-225 in Appendix H-11 demonstrate the relationships between data for various SVOCs/PAHs in SWBZ groundwater and FWBZ groundwater in the landfill area. AWQC and Alameda Point background values are not available for SVOCs/PAHs in groundwater.

VOCs

Only six VOCs were detected in SWBZ groundwater samples collected from the landfill area of the site. Acetone was detected in 30%, chlorobenzene was detected in 20%, ethyl benzene was detected in 10%, styrene was detected in 10%, tert-butyl alcohol was detected in 40%, and toluene was detected in 10% of the landfill area SWBZ groundwater samples analyzed for VOCs.

All VOCs detected in landfill area SWBZ groundwater exhibited maximum concentrations well below Region 9 PRGs. The majority of detected VOC concentrations were only slightly above the detection limit. Overall, the number of individual VOCs detected and the mean and maximum detected concentrations of individual VOCs were appreciably lower in landfill area SWBZ groundwater compared to landfill-area FWBZ groundwater. These data do not suggest that VOCs represent a contamination issue in the landfill-area SWBZ.

VOC data for SWBZ groundwater samples from the site are presented on Figure E-17 in Appendix E. Bubble plots showing the concentrations of VOCs in SWBZ groundwater samples collected from the site are presented as Figures F-577 through F-582 in Appendix F-27. The box and whisker plots presented as Figures H-258 through H-286 in Appendix H-14 demonstrate the relationships between data for various VOCs in landfill area SWBZ groundwater and FWBZ groundwater. AWQC and Alameda Point background values are not available for VOCs in groundwater.

Petroleum Hydrocarbons

DRO and RRO were detected in 50% and 10%, respectively, of the groundwater samples collected from the SWBZ in the landfill area of IR Site 2. GRO were not detected in any SWBZ groundwater samples collected from the landfill area.

The maximum concentrations of DRO and RRO detected in landfill area SWBZ groundwater samples were 600 and 450 µg/L, respectively. DRO and RRO were detected only in the northern portion of the landfill. Mean and maximum detected concentrations of DRO and RRO were significantly lower for landfill-area SWBZ groundwater compared to landfill area FWBZ groundwater. The landfill-area SWBZ petroleum hydrocarbon concentrations are low, and, although these compounds also were detected in FWBZ groundwater in the same general area, do not appear to suggest any localized hot spots of petroleum hydrocarbons or any contaminant source areas in the landfill-area SWBZ.

TPH data for SWBZ groundwater samples from the site are presented on Figure E-16 in Appendix E. Bubble plots showing the concentrations of petroleum hydrocarbons in SWBZ groundwater samples collected from the site are presented as Figures F-583 and F-584 in Appendix F-28. The box and whisker plots presented as Figures H-287 through H-289 in Appendix H-15 demonstrate the relationships between data for various petroleum hydrocarbons in landfill area FWBZ groundwater and SWBZ groundwater. There are no specific Alameda Point background values, PRGs, or AWQC for TPHs in groundwater.

Radionuclides

Gross beta was detected in 86%, Ra-228 was detected in 57%, and gross alpha was detected in 14% of the seven landfill area SWBZ groundwater samples analyzed for these compounds. The maximum concentrations of gross alpha, gross beta, and Ra-228 detected in landfill area SWBZ groundwater were 52 pCi/L, 127 pCi/L, and 5.9 pCi/L, respectively. Tritium was not detected in any of the landfill area SWBZ groundwater samples analyzed for this compound.

Those radionuclides detected in both landfill-area SWBZ and FWBZ groundwater (i.e., gross beta and Ra-228) exhibited relatively consistent mean and maximum detected concentrations. Radionuclides also were detected in landfill soils. However, the number of radionuclides detected in SWBZ groundwater was very low, the concentrations of radionuclides detected in landfill area SWBZ generally were very low, and overall there do not appear to be any radionuclide hotspots in landfill-area SWBZ groundwater.

Radionuclide data for SWBZ groundwater samples from the site are presented on Figure E-14 in Appendix E. Bubble plots showing the concentrations of radionuclides in SWBZ groundwater samples collected from the site are presented as Figures F-585 through F-587 in Appendix F-29. The box and whisker plots presented as Figures H-290 through H-297 in Appendix H-16 demonstrate the relationships between data for various radionuclides in landfill area FWBZ groundwater and SWBZ groundwater. There are no specific background values, PRGs, or AWQC values for radionuclides in groundwater.

General Chemistry

Several landfill area SWBZ groundwater samples were analyzed for alkalinity, chloride, nitrate, nitrite, sulfate, and sulfide. Chloride was detected in 100% of the samples and sulfate was detected in 57% of the samples. The maximum concentration for chloride and sulfate are 16,000 and 2,200 mg/L, respectively. Detections were only found in wells located in the southern portion of the landfill

for both chloride and sulfate. These analytical parameters are, in and of themselves, not indicative of any type or distribution of contamination in landfill area SWBZ groundwater, and therefore are not discussed here in detail.

Overall, the relative lack of contamination in the landfill area SWBZ is likely supported, in part, by the existence of the BSU/Young Bay Mud confining unit beneath the FWBZ across much of the site.

5.3.2 Wetlands

The following subsections describe the most likely sources of contamination in the wetland portion of the site, and describe the comprehensive CSM of site contamination developed for the wetland portion of the site by environmental medium (i.e., surface and subsurface soil, groundwater, surface water and sediment) and analyte class (e.g., metals, VOCs, and pesticides).

5.3.2.1 Potential Sources of Contamination

As indicated in Section 2.0, historical information suggests that very limited waste disposal activities may have occurred in the wetlands. Specifically, historical information indicates that waste placement eventually extended to most of the northern and eastern areas of the landfill and part of the northern wetland pond.

In addition, to aid in identifying potential contamination source areas (i.e., hot spots) within the wetland, historical information from the IAS (E&E, 1983) was used to develop a map identifying discrete wastes that were disposed at IR Site 2. The locations of these discrete waste disposal areas were mapped during the IAS. The discrete disposal areas suspected to be in the wetlands on the basis of historical information are listed below, and are summarized in greater detail in Section 2.11 and shown on Figure 2-10.

- **Dredge Spoil Spreading Area:** The southern portion of the site served as a location for the deposition of dredge spoils from Oakland Inner Harbor, the pier area and turning basin of Alameda Point, and Seaplane Lagoon.

- **Scrap Metal Disposal Area:** Scrap metal from PWC sites was reportedly disposed of in the wetland area.

As indicated in Section 5.2.2, a geophysical survey was successful in identifying the presence and general extent of subsurface geophysical anomalies at the site. In general, no geophysical anomalies were identified in the wetland area. A small anomaly in the far southwestern corner of the wetlands could represent historical scrap metal disposal (see above and see Section 2.0), but the identified anomaly was highly localized and the particular waste type could not be determined. The geophysical survey could not, in and of itself, specifically confirm or refute the existence of the specific historical waste disposal areas described above and in Section 2.0. No exploratory trenching was completed in the wetland area during the RI.

5.3.2.2 Wetland Surface Soils

During the course of the RI at IR Site 2, a total of 45 surface soil samples (i.e., from the 0 to 1 ft bgs interval) were collected from the wetland area of the site. Samples were generated during the dry and wet

season soil sampling programs. The following analytical parameters were evaluated in the surface soil samples collected from the wetland area:

- All 45 surface soil samples were analyzed for pesticides, PCBs, and moisture content;

- 44 surface soil samples were analyzed for metals, VOCs, SVOCs (including PAHs), and moisture content;

- 19 surface soil samples were analyzed for hexavalent chromium and TPHs;

- 10 surface soil samples were analyzed for TOC and grain-size distribution;

- 7 surface soil samples were analyzed for TBT; and,

- 4 surface soil samples were analyzed for PCDDs/PCDFs, gross alpha, gross beta, Ra-226, Ra-228, Pb-210, U-234, U-235, and U-238.

Table 5-8 provides a general statistical summary of the surface soil data generated in the wetland portion of the site during the RI.

Metals

Aluminum, arsenic, barium, beryllium, chromium, cobalt, copper, iron, lead, magnesium, manganese, mercury, nickel, silver, vanadium, and zinc were detected in all of the surface soil samples collected from the landfill. This is nearly identical to the list of metals that were detected in 100% of the surface soil samples collected from the landfill portion of the site. Selenium and thallium were detected in 84% or more of the surface soil samples. Antimony and cadmium were detected in roughly 58% and 66% of the surface soil samples from the wetland area, respectively. Molybdenum was not detected in any of the surface soil samples collected. TBT was detected in approximately 30% of the surface soil samples analyzed for this compound, and hexavalent chromium was detected in only approximately 10% of the surface soil samples analyzed for this metal (i.e., only two of the 19 samples analyzed for this metal). These frequencies of detection are also highly consistent with conditions observed for surface soils in the landfill portion of the site.

Widespread exceedances of established Alameda Point background values for metals were identified in surface soils across the wetland for aluminum, arsenic, barium, beryllium, cadmium, chromium, cobalt, copper, iron, lead, magnesium, manganese, mercury, nickel, vanadium, and zinc. TBT was detected at low concentrations at two wetland area surface soil sampling locations, with the highest detected concentration occurring in the northwestern portion of the wetland. When detected, antimony, hexavalent chromium, molybdenum, selenium, silver, and thallium were detected only at very low concentrations, with none of the silver concentrations exceeding the applicable background value. Alameda Point background values have not been developed for TBT, hexavalent chromium, molybdenum, selenium, or thallium in soils. Arsenic, iron, and vanadium were detected at several wetland surface soil sampling locations at concentrations exceeding Region 9 PRGs (note that the PRG for arsenic is lower than the established Alameda Point background value for this metal).

In general, there is no obvious pattern in the occurrence of metals concentrations in wetland surface soils that would suggest localized hot spots of metals contamination or clear source areas. Metals concentrations detected in surface soils in the wetland portion of the site are generally consistent with metals concentrations detected in surface soil samples collected from the landfill portion of the site. The mean detected concentrations of the following metals were actually higher in wetland surface soils compared to landfill surface soils: aluminum; arsenic; beryllium; chromium; cobalt; iron; magnesium; manganese; nickel; selenium; thallium; vanadium; and TBT. In addition, the maximum detected concentrations of the following metals were higher in wetland area surface soils relative to

landfill surface soils: aluminum; beryllium; chromium; cobalt; iron; lead; magnesium; manganese; nickel; selenium, thallium; and vanadium.

Metals concentrations detected in surface soils in the wetland portion of the site are also generally consistent with metals concentrations detected in reference surface soil samples collected from the wetland portion of CCSP. The mean detected concentrations of the following metals in surface soils were actually higher in the CCSP wetland relative to the wetland portion of the site: aluminum; arsenic; beryllium; chromium; cobalt; copper; iron; magnesium; manganese; nickel; selenium; thallium; vanadium; and zinc. In addition, the maximum detected concentrations of the following metals in surface soils were actually higher in the CCSP wetland relative to the wetland portion of the site: aluminum; manganese; and vanadium. Reference surface soil samples from the CCSP wetland were not analyzed for hexavalent chromium or TBT.

Metals data for soil samples from the site are presented on Figure E-1 in Appendix E. Bubble plots showing the concentrations of metals in surface soil samples collected from the site during the RI are presented as Figures F-1 through F-23 in Appendix F-1. The box and whisker plots presented as Figures H-1 through H-23 in Appendix H-1 demonstrate the relationships between data for various metals in wetland area surface soils and metals data associated with CCSP upland reference surface soils. These box and whisker plots also demonstrate the relationships between data for various metals in landfill area surface soils and wetland area surface soils.

SVOCs/PAHs

The following SVOCs were not detected in any surface soil samples collected from the wetland portion of the site: 1,2,4,5-tetrachlorobenzene; 2,4,5-trichlorophenol, 2,4,6-trichlorophenol; 2,4-dichlorophenol; 2.4-dimethylphenol; 2,4-dinitrophenol; 2-chloronaphthalene; 2-chlorophenol; 2-methyl-4,6-dinitrophenol; 2-methylphenol; 2-nitrophenol; 2-nitroaniline; 3,3-dichlorobenzidine; 3-nitroaniline; 4-bromophenyl phenyl ether; 4-chlorophenyl phenyl ether; 4-methylphenol; 4-nitroaniline; 4-nitrophenol; atrazine; *bis*(2-chloroethoxy)methane; *bis*(2-chloroethyl)ether; *bis*(chloroisopropyl)ether; dimethyl phthalate; di-*n*-octyl phthalate; hexachlorobenzene; hexachlorobutadiene; hexachlorocyclopentadiene; hexachloroethane; isophorone; *n*-nitrosodi-*n*-propylamine; *n*-nitrosodiphenylamine; and pentachlorophenol.

A number of SVOCs/PAHs were detected in the surface soil samples collected from the wetland portion of the site. The following SVOCs were detected in 5% or less of the surface soil samples collected from the landfill portion of the site: 4-chloro-3-methylphenol; caprolactam; and diethyl phthalate. The following SVOCs were detected in roughly 11% to 36% of the surface soil samples collected from the wetland area: 1,1-biphenyl; acetophenone; benzaldehyde; carbazole; di-*n*-butyl phthalate; and phenol. 2-methylnaphthalene, acenaphthene, acenaphthylene, anthracene; *bis*(2-ethylhexyl) phthalate, dibenz(a,h)anthracene, dibenzofuran, fluorene, and naphthalene were detected in approximately 41% to 77% of the surface soil samples collected from the wetland. Benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(g,h,i)perylene, benko(k)fluoranthene, butyl benzyl phthalate, chrysene, fluoranthene, indeno(1,2,3-cd)pyrene, phenanthrene, and pyrene were all detected in 80% to 100% of the surface soil samples collected from the landfill. These frequencies of detection are highly consistent with those observed in the landfill area surface soil samples.

Overall, low detected concentrations of SVOCs/PAHs in surface soils occur widespread over the majority of the wetland portion of the site. There does not appear to be a clear pattern of SVOC/PAH occurrence in surface soils in the wetland portion of the site. For all SVOCs/PAHs for which PRGs exist, with the exception of benzo(a)pyrene, the maximum detected analyte concentration in wetland

surface soils is well below the PRG. The maximum detected concentration of benzo(a)pyrene (140 µg/kg) represents an exceedance of the relevant PRG (62.1 µg/kg). Benzo(a)pyrene concentrations exceeding the relevant PRG by a small margin are relatively widespread in wetland area surface soils and do not appear to indicate a clear pattern that would suggest localized sources of contamination or contaminant hot spots.

For all SVOCs/PAHs that were detected in both landfill area and wetland area surface soil samples, both mean and maximum individual compound concentrations were higher in the landfill area surface soils compared to the wetland area. For certain compounds, mean and maximum concentrations were relatively consistent between the landfill and wetland area surface soils. However, in general, the mean and maximum concentrations of various SVOC/PAH compounds were significantly lower in the wetland area surface soils.

Reference surface soil samples were collected from the CCSP wetland for the analysis of SVOCs/PAHs. Alrthough several SVOCs/PAHs were detected in the reference surface soil samples, the number of individual SVOCs/PAHs detected in the CCSP surface soils was significantly less than the number of individual SVOC/PAH compounds detected in surface soils from the wetland portion of the site. The maximum concentrations of all SVOCs/PAHs detected in surface soils from both the site wetland and the CCSP wetland were appreciably lower for the CCSP soils. SVOCs/PAHs were not detected in surface soil samples from the CCSP wetland at concentrations exceeding relevant PRGs.

Specific background values have been developed for certain SVOC/PAH compounds in soils at Alameda Point. For all SVOC/PAH compounds for which an Alameda Point background value exists (benzo(a)pyrene, benzo(g,h,i)perylene, chrysene, fluoranthene, indeno(1,2,3-cd)pyrene, phenanthrene, and pyrene), the maximum detected concentrations in surface soils from the wetland portion of the site are well below the established background level.

SVOC/PAH data for soil samples from the site are presented on Figure E-2 in Appendix E. Bubble plots showing the concentrations of SVOCs/PAHs in surface soil samples collected from the site during the RI are presented as Figures F-24 through F-64 in Appendix F-2. The box and whisker plots presented as Figures H-24 through H-66 in Appendix H-2 demonstrate the relationships between data for various SVOCs/PAHs in wetland area surface soils and SVOC/PAH data associated with CCSP wetland reference surface soils. These box and whisker plots also demonstrate the relationships between data for various SVOCs/PAHs in wetland area surface soils and landfill area surface soils.

PCBs

Aroclors 1016 and 1221 were not detected in any of the surface soil samples collected from the wetland area of the site that were analyzed for PCBs. Aroclor 1242 was detected in only 2% (i.e., one of the 45 samples analyzed) and Aroclor 1232 was detected in only 5% (i.e., two of the 45 samples analyzed) of the wetland area surface soil samples that were analyzed for PCBs. The one detected concentration of Aroclor 1242 in wetland area surface soils was only approximately 0.2 mg/kg, and the maximum detected concentration of Aroclor 1232 was only approximately 0.6 mg/kg.

Aroclor 1248 was detected in approximately 9% and Aroclor 1260 was detected in approximately 11% of the surface soil samples collected from the wetland area of the site that were analyzed for PCBs, while Aroclor 1254 was detected in approximately 66% of the wetland area surface soil samples. The maximum detected concentrations of Aroclors 1248, 1254, and 1260 were approximately 0.5 mg/kg, 0.4 mg/kg, and 0.2 mg/kg, respectively. Low detected concentrations of

individual Aroclors were observed across the wetland portion of the site, with generally no clear evidence suggesting the presence of discrete contamination hot spots or sources. Aroclor 1254 was detected in one wetland area surface soil sample exceeding the Region 9 PRG, at a location between the two ponds immediately adjacent to the central portion of the landfill. For all PCBs that were detected in both landfill area and wetland area surface soil samples, both mean and maximum individual Aroclor concentrations were higher in the landfill area surface soil dataset compared to the wetland area.

Using the NOAA method, total PCBs were calculable for 95% of the surface soil samples in the wetland portion of the site. Total PCB concentrations calculated using this method for wetland area surface soils were all less than 0.5 mg/kg. The maximum total PCB concentration calculated using the NOAA method for wetland area surface soils (0.47 mg/kg) represents only a minimal exceedance of the Region 9 PRG (0.22 mg/kg). NOAA-calculated total PCB concentrations in surface soils exceed the Region 9 PRG for only three locations in the wetland portion of the site, and these three sampling locations generally were very close to the landfill boundary in the northern portion of the wetlands. Using the sum of Aroclors, total PCBs were calculated for 80% of the wetland area surface soil samples, with a maximum concentration of approximately 1.5 mg/kg. Total PCB concentrations derived from the sum of Aroclors in wetland surface soils exceed the Region 9 PRG for seven locations (two of which were the same as locations exhibiting Region 9 PRG exceedances in the NOAA-based total PCB dataset), and these sampling locations also were generally close to the landfill boundary in the northern portion of the wetlands.

There are no specific background values for PCB Aroclors or total PCBs in soils at Alameda Point. However, reference surface soil samples were collected from the CCSP wetland for analysis of PCBs. No PCB Aroclors were detected in the CCSP wetland reference soil samples. Total PCB concentrations calculated for the CCSP reference wetland samples (using the NOAA method only, as no Aroclors were detected) generally were appreciably lower than total PCB concentrations calculated for wetland surface soils at IR Site 2.

Aroclor and total PCB data for soil samples from the site are presented on Figure E-3 in Appendix E. Bubble plots showing the concentrations of PCB Aroclors and total PCBs in surface soil samples collected from the site during the RI are presented as Figures F-65a through F-72c in Appendix F-3. The box and whisker plots presented as Figures H-67a through H-74c in Appendix H-3 demonstrate the relationships between data for various Aroclors and total PCBs in wetland area surface soils and landfill area surface soils. These box and whisker plots also demonstrate the relationships between PCB data from wetland area surface soils and CCSP wetland reference surface soil data.

Pesticides

alpha-BHD, aldrin, *delta*-HCH, endosulfan I, endosulfan sulfate, endrin, endrin aldehyde, endrin ketone, heptachlor, and toxaphene were not detected in any of the surface soil samples from the wetland area of the site that were evaluated for pesticides. *beta*-BHC and heptachlor epoxide were detected in 5% or fewer of all surface soil samples from the wetland area. 2,4-DDE, *gamma*-BHC (lindane), and methoxyclor were detected in approximately 7% to 11% of the wetland area surface soil samples analyzed for pesticides. 2,4-DDD, 2,4-DDT, 4,4-DDT, *alpha*-chlordane, *cis*-nonachlor, dieldrin, endosulfan II, *gamma*-chlordane, and *trans*-nonachlor were detected in approximately 38% to 80% of the surface soil samples from the wetland that were analyzed for pesticides. 4,4-DDD and 4,4-DDE were detected in 87% or more of the surface soil samples from the wetland that were analyzed for pesticides. These frequencies of detection are generally consistent with the detections of pesticides in landfill area surface soil samples.

Maximum detected concentrations of individual pesticides in wetland area surface soils generally were below 10 μg/kg. For all pesticides detected in wetland area surface soils for which there exists a Region 9 PRG, the maximum detected concentration at the site was well below the relevant PRG. Very low concentrations of pesticides were detected across the wetland portion of the site, and there is no clear pattern that would suggest the presence of contamination hot spots or sources.

For all pesticides that were detected in both landfill area and wetland area surface soil samples, mean and maximum individual pesticide concentrations generally were somewhat consistent. However, both mean and maximum individual pesticide concentrations were higher in the landfill area surface soil dataset compared to the wetland area for all individual compounds, with the exception of dieldrin. For dieldrin, the maximum detected concentration in wetland surface soils (10.5 μg/kg) was slightly higher than the maximum concentration detected in landfill area surface soils (6.2 μg/kg).

Reference surface soil samples were collected from the CCSP wetland for the analysis of pesticides. Although several individual pesticides were detected in the reference surface soil samples from the CCSP wetland, the number of pesticides detected in the CCSP surface soils was slightly less than the number of individual pesticides detected in surface soils from the wetland portion of the site. Maximum concentrations of those pesticides detected in surface soils from both the site wetland and the CCSP wetland generally were relatively consistent or lower for the CCSP soils. The exception to this was 4,4-DDD, for which the maximum detected concentration in CCSP wetland surface soils exceeded the maximum detected concentration from site wetland surface soils by a factor of approximately four. There are no specific background values for pesticides in soils at Alameda Point.

Pesticide data for soil samples from the site are presented on Figure E-3 in Appendix E. Bubble plots showing the concentrations of pesticides in surface soil samples collected from the site during the RI are presented as Figures F-73 through F-93 in Appendix F-4. The box and whisker plots presented as Figures H-75 through H-103 in Appendix H-4 demonstrate the relationships between data for various pesticides in wetland area surface soils and pesticide data associated with CCSP wetland reference surface soils, in addition to relevant PRGs. These box and whisker plots also demonstrate the relationships between data for various pesticides in wetland area surface soils and landfill area surface soils.

VOCs

Only the following VOCs were detected in any of the surface soil samples collected from the wetland portion of the site that were analyzed for these compounds: methyl ethyl ketone (MEK); acetone; chlorobenzene; cyclohexane; MTBE; methylcyclohexane; and methylene chloride. Acetone was detected in approximately 7% of the surface soil samples analyzed for VOCs, and methylene chloride was detected in roughly 43% of the surface soil samples analyzed for VOCs. The maximum detected concentrations of both of these VOCs were very low (i.e., 23 μg/kg for acetone and 73 μg/kg for methylene chloride) and both are common laboratory contaminants.

MEK was detected in only one surface soil sample from the wetland at a concentration of 420 μg/kg. Chlorobenzene also was detected in only one surface soil sample from the wetland at a concentration of 4.7 μg/kg. Cyclohexane, MTBE, and methylcyclohexane were detected in approximately 7% to 14% of the wetland area surface soil samples analyzed for VOCs. The maximum detected concentrations of these three VOCs were very low (6.9 μg/kg, 3.8 μg/kg and 14 μg/kg, respectively). The detected concentrations of all VOCs in wetland area surface soils were well below the respective PRGs for these compounds. None of the VOC detections in wetland area surface soils appears to represent a discrete source area or evidence of any substantial VOC contamination in surface soil in the wetland.

Only MEK, acetone, and methylene chloride were detected in surface soil samples from both the wetland and landfill portions of the site. The mean and maximum concentrations of acetone and methylene chloride generally were consistent but slightly higher for the landfill area surface soil dataset compared to the wetland surface soils. As indicated above, both acetone and methylene chloride are common laboratory contaminants. The mean and maximum concentrations of MEK were actually higher in the wetland area surface soil dataset compared to the landfill area surface soils. However, MEK was detected only in one surface soil sample each from the wetland and the landfill areas.

Specific background values have not been developed for VOCs in soils at Alameda Point. Reference surface soil samples were collected from the CCSP wetland for analysis of VOCs, but VOCs were not detected in these reference surface soil samples from the CCSP wetland.

VOC data for soil samples from the site are presented on Figure E-4 in Appendix E. Bubble plots showing the concentrations of VOCs in surface soil samples collected from the site during the RI are presented as Figures F-94 through F-101 in Appendix F-5. The box and whisker plots presented as Figures H-104 through H-127 in Appendix H-5 demonstrate the relationships between data for VOCs in wetland area surface soils and landfill area surface soils.

PCDD/PCDF

Several individual PCDD/PCDF congeners were detected in most of the surface soil samples collected from the wetland area of the site that were analyzed for these compounds. In accordance with common practice and guidance from U.S. EPA, the individual PCDD/PCDF congeners detected in discrete surface soil samples were summed as described in Section 5.1 to generate a TEQ for each sampling point.

For the majority of the wetland area surface soil sampling locations where PCDDs/PCDFs were analyzed, the calculated TEQ marginally exceeds the Region 9 PRG (i.e., the PRG for 2,3,7,8-TCDD, the most toxic of the PCDD/PCDF congeners). Overall, it appears that somewhat elevated levels of PCDDs/PCDFs are widespread in surface soils across the wetland portion of the site. However, the number of sampling locations that were evaluated for PCDD/PCDFs was limited in the wetland portion of the site.

For all PCDDs/PCDFs that were detected in both landfill area and wetland area surface soil samples, mean and maximum individual congener concentrations generally were significantly higher for the landfill area surface soils. Mean and maximum TEQs calculated for landfill area surface soil sampling locations also generally were appreciably higher for the landfill area surface soil dataset.

Reference surface soil samples were collected from the wetland portion of CCSP, but were not analyzed for PCDD/PCDF. However, specific sampling was completed during the RI to evaluate the potential presence and extent of PCDD/PCDF contamination at Alameda Point reference locations (i.e., locations on Alameda Point but not affected in any known way by historical industrial or waste disposal activities). Specifically, soil sampling was completed at the same Alameda Point reference locations as were assessed during the radiological survey program (see Section 5.2.3 and TTFW, 2005). Although the PCDD/PCDF reference sampling was limited to upland locations, it provides a valuable basis of comparison for the wetland area surface soil sampling conducted during the RI. Several individual PCDD/PCDF congeners were detected in most of the surface soil samples collected from the Alameda Point reference locations. The concentrations of individual PCDD/PCDF congeners detected in the Alameda Point reference samples generally were consistent with those detected in surface soil samples collected from the wetland portion of the site. In some cases, the

concentrations detected in wetland area surface soil samples were somewhat higher. Similarly, TEQs calculated for the sampling locations in the Alameda Point reference areas generally were consistent with but lower than TEQs calculated for surface soil sampling locations in the wetland portion of the site. No calculated TEQs for reference area surface soil samples exceeded the Region 9 PRG for 2,3,7,8-TCDD.

TEQs calculated for soil samples from the site are presented on Figure E-3 in Appendix E. Figure F-102 in Appendix F-6 is a bubble plot of TEQs generated for surface soil samples collected from the site during the RI. The box and whisker plots presented as Figures H-128 through H-146 in Appendix H-6 demonstrate the relationships between data for various PCDDs/PCDFs and TEQs in wetland area surface soils and the reference PCDD/PCDF and TEQ data developed for Alameda Point. These box and whisker plots also demonstrate the relationships between data for various PCDDs/PCDFs and TEQs in wetland area surface soils and landfill area surface soils.

Petroleum Hydrocarbons

RRO was detected in all of the surface soil samples collected from the wetland area that were analyzed for these compounds. DRO was detected in roughly 58% of all surface soil samples collected from the wetland area that were analyzed for these compounds. GRO was not detected in any of the surface soil samples collected from the wetland area that were analyzed for petroleum hydrocarbons. The maximum concentrations of DRO and RRO detected in wetland area surface soil samples were 86 mg/kg and 330 mg/kg, respectively. These maximum concentrations are significantly lower than concentrations of DRO and RRO detected in landfill area surface soil samples. Mean detected concentrations of DRO and RRO also were significantly lower for the wetland area surface soil samples in comparison to the landfill area surface soil samples.

Low detected concentrations of DRO and GRO appear to be relatively widespread across the wetland area of the site, and do not suggest the presence of localized contamination hot spots or sources.

Specific background values have not been developed for TPHs in soils at Alameda Point. TPHs were not analyzed for in the reference surface soil samples collected from the CCSP wetland. In addition, there are no PRGs for DRO or RRO.

TPH data for soil samples from the site are presented on Figure E-3 in Appendix E. Bubble plots showing the concentrations of petroleum hydrocarbons in surface soil samples collected from the site during the RI are presented as Figures F-103 through F-105 in Appendix F-6. The box and whisker plots presented as Figures H-147 through H-149 in Appendix H-7 demonstrate the relationships between data for petroleum hydrocarbons in wetland area surface soils and landfill area surface soils.

Radionuclides

Lead 210 was detected in only the duplicate of one of the wetland area surface soil samples evaluated for this analyte, at an estimated concentration of 1.6 pCi/g. U-235 was detected in 3 of the 4 wetland area surface soil samples that were analyzed for radionuclides, with a maximum detected concentration of 0.04 pCi/g. Gross alpha, gross beta, Ra-226, Ra-228, U-234, and U-238 were all detected in all of the wetland area surface soil samples that were analyzed for radionuclides. The maximum concentrations of the radionuclides detected in wetland area surface soils were: gross alpha (15.3 pCi/g); gross beta (38 pCi/g); Ra-226 (0.52 pCi/g); Ra-228 (1.13 pCi/g); U-234 (0.65 pCi/g); and U-238 (0.58 pCi/g). Concentrations of Ra-226 were detected slightly above the average background value developed during the radiological survey conducted in 2004 (see Section 5.2.3 and

TTFW, 2005), but within the range of concentrations that would be expected from background conditions.

There does not appear to be a trend or pattern in the occurrence of radionuclides in wetland area surface soils that would suggest the presence of contamination hot spots or localized sources. As discussed in Section 4.0, the number of samples collected for analysis of radionuclides was limited in the wetland area. However, based on the limited amount of sampling conducted for radionuclides in the wetland area, the highest Ra-226 concentrations detected in wetland area surface soil samples appear to be isolated to the central portion of the wetland. Overall, the concentrations of radionuclides detected in the wetland surface soils generally were consistent with concentrations of radionuclides detected in the landfill area surface soils.

Specific background values have not been developed for radionuclides in soils at Alameda Point other than the value generated by the Navy during its recent survey program (see Section 5.2.3). Radionuclides were not analyzed for in the reference surface soil samples collected from the CCSP wetland. In addition, there are no PRGs for radionuclides.

Radionuclide data for soil samples from the site are presented on Figure E-1 in Appendix E. Bubble plots showing the concentrations of radionuclides in surface soil samples collected from the site during the RI are presented as Figures F-106 through F-113 in Appendix F-7. The box and whisker plots presented as Figures H-150 through H-157 in Appendix H-8 demonstrate the relationships between data for radionuclides in wetland area surface soils and landfill area surface soils.

General Chemistry

Several surface soil samples collected from the wetland portion of the site were evaluated for moisture content, TOC, and grain-size distribution. These analytical parameters are, in and of themselves, not indicative of any type or distribution of contamination in wetland area surface soils, and therefore are not discussed here in detail.

5.3.2.3   Wetland Subsurface Soils

During the course of the RI at IR Site 2, a total of 16 subsurface soil samples (i.e., from depths below 1 ft bgs) were collected from the wetland area of the site. Samples were generated during the dry season soil sampling programs. The following analytical parameters were evaluated in the subsurface soil samples collected from the wetland area:

- All 16 subsurface soil samples were analyzed for metals, SVOCs (including PAHs), VOCs, PCBs, pesticides, and moisture content;

- 10 subsurface soil samples were analyzed for hexavalent chromium;

- 8 subsurface soil samples were analyzed for TOC and grain-size distribution;

- 6 subsurface soil samples were analyzed for TBT; and,

- 4 subsurface soil samples were analyzed for TPHs, PCDDs/PCDFs, gross alpha, gross beta, Pb-210, Ra-226, Ra-228, U-234, U-235, and U-238.

Table 5-9 provides a general statistical summary of the subsurface soil data generated in the wetland portion of the site during the RI.

Metals

Aluminum, arsenic, barium, beryllium, chromium, cobalt, copper, iron, lead, magnesium, manganese, mercury, nickel, silver, vanadium, and zinc were detected in 100% of the subsurface soil samples collected from the wetland. Antimony, cadmium, selenium, and thallium were detected in 50% or more of the subsurface soil samples. Molybdenum was detected in roughly 6% of the subsurface soil samples collected from the wetland area (i.e., one of the 16 subsurface soil samples). TBT was detected in approximately 33% of the subsurface soil samples analyzed for this compound (i.e., 2 of the 6 samples analyzed for this compound), and hexavalent chromium was detected in 10% of the subsurface soil samples analyzed for this metal (i.e., one of the 10 samples analyzed for this metal). In general, the frequencies of detection of individual metals in wetland area subsurface soils is consistent with the frequencies of detection of these metals in surface soils from the wetland area and subsurface soils from landfill area.

Exceedances of established Alameda Point background values for metals were identified in subsurface soils in the wetland area for aluminum, antimony, arsenic, barium, beryllium, cadmium, chromium, cobalt, copper, iron, lead, magnesium, manganese, mercury, nickel, silver, vanadium, and zinc. In general, there is no obvious pattern in the occurrence of these metals in wetland area subsurface soils that would suggest localized hot spots of contamination or clear contaminant source areas. When detected, hexavalent chromium, molybdenum, selenium, thallium, and TBT were detected only at low concentrations that do not exceed Region 9 PRGs (note there are no established Alameda Point background values for these metals). Iron and vanadium were detected in a small number of wetland area subsurface soils at concentrations exceeding applicable PRGs. Arsenic was detected in subsurface soils across the wetland at concentrations exceeding the applicable PRG (note that the PRG for arsenic is lower than the established Alameda Point background value for this metal).

For all metals that were detected in both wetland area subsurface and surface soils, mean and maximum detected concentrations were highly consistent. Comparing metals concentrations in wetland area subsurface soils and landfill area subsurface soils yields different results. The mean and maximum detected concentrations of the following metals were appreciably higher for landfill area subsurface soils compared to wetland area subsurface soils: antimony; cadmium; copper; lead; zinc; and TBT. In addition, the maximum detected concentrations (but not the mean) of the following metals were appreciably higher for landfill area subsurface soils compared to wetland area subsurface soils: aluminum; barium; chromium; iron; mercury; molybdenum; and silver.

Reference subsurface soil samples were not collected from the CCSP wetland for the analysis of metals. However, reference surface soil samples collected from the CCSP wetland provide a valuable basis for comparison. Metals concentrations detected in subsurface soils in the wetland portion of the site are generally consistent with metals concentrations detected in reference surface soil samples collected from the wetland portion of CCSP. The mean detected concentrations of the following metals were actually higher in CCSP wetland surface soils relative to subsurface soils in the wetland portion of the site: aluminum; beryllium; chromium; cobalt; copper; iron; magnesium; manganese; nickel; selenium; thallium; and vanadium. In addition, the maximum detected concentrations of the following metals in surface soils from the CCSP wetland were actually higher relative to subsurface soils from the wetland portion of the site: aluminum; iron; magnesium; manganese; selenium; and vanadium. Reference surface soil samples from the CCSP wetland were not analyzed for hexavalent chromium or TBT.

Metals data for subsurface soil samples from the site are presented on Figures E-5 and E-6 in Appendix E. Bubble plots showing the concentrations of metals in subsurface soil samples collected

from the site during the RI are presented as Figures F-116 through F-161 in Appendix F-9. Bubble plots were generated for both the 1 to 4 ft bgs interval and the interval below 4 ft bgs. The description of the nature and extent of metals in wetland area subsurface soils above integrates the entire subsurface zone below 1 ft bgs. The box and whisker plots presented as Figures H-1 through H-23 in Appendix H-1 demonstrate the relationships between data for various metals in wetland area subsurface soils and metals data associated with CCSP wetland reference surface soils. These box and whisker plots also demonstrate the relationships between data for various metals detected in wetland area subsurface soils, wetland area surface soils, and landfill area subsurface soils.

SVOCs/PAHs

Only the following SVOCs/PAHs were detected in any subsurface soil samples collected from the wetland portion of the site: 1,1-biphenyl; 2-methylnaphthalene; 4-chloroaniline; acenaphthene; acenaphthylene; acetophenone; anthracene; benzaldehyde; benzo(a)anthracene; benzo(a)pyrene; benzo(b)fluoranthene; benzo(g,h,i)perylene; benzo(k)fluoranthene; *bis*(2-ethylhexyl) phthalate; butyl benzyl phthalate; carbazole; chrysene; dibenz(a,h)anthracene; dibenzofuran; diethyl phthalate; di-*n*-butyl phthalate; fluoranthene; fluorene; indeno(1,2,3-cd)pyrene; naphthalene; pentachlorophenol; phenanthrene; and pyrene. Of these compounds, 1,1-biphenyl, 4-chloroaniline, benzaldehyde, diethyl phthalate, and pentachlorophenol were detected in no more than one sample.

2-methylnaphthalene, acenaphthene, acenaphthylene, acetophenone, anthracene, benzo(k)fluor-anthene, *bis*(2-ethylhexyl) phthalate, carabazole, dibenz(a,h)anthracene, dibenzofuran, di-*n*-butyl phthalate, fluorene, naphthalene, and pentachlorophenol were detected in approximately 19% to 75% of wetland area subsurface soil samples. Benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(g,h,i)perylene, butyl benzyl phthalate, chrysene, fluoranthene, indeno(1,2,3-cd)pyrene, phenanthrene, and pyrene were all detected in approximately 81% to 100% of the subsurface soil samples collected from the wetland.

Overall, detected concentrations of SVOCs/PAHs in subsurface soils occur widespread over the majority of the wetland portion of the site. However, detected SVOC/PAH concentrations in wetland area subsurface soils are generally very low and there does not appear to be a clear pattern of SVOC/PAH occurrence that would suggest localized sources of contamination. For all SVOCs/PAHs for which a Region 9 PRG exists, with the exception of benzo(a)pyrene, the maximum detected analyte concentration in wetland subsurface soils is well below the PRG value. The maximum detected concentration of benzo(a)pyrene in wetland area subsurface soils (i.e., 98 µg/kg) represents only a marginal exceedance of the applicable PRG (i.e., 62.1 µg/kg).

Mean and maximum concentrations of individual SVOC/PAH compounds detected in wetland area subsurface soils generally were highly consistent with the mean and maximum concentrations of SVOC/PAH compounds detected in wetland area surface soils. Mean and maximum concentrations of individual SVOC/PAH compounds detected in wetland area subsurface soils generally were significantly lower compared to mean and maximum concentrations of SVOC/PAH compounds detected in landfill area subsurface soils.

Reference subsurface soil samples were not collected from the CCSP wetland. However, reference surface soil samples were collected from the CCSP wetland for the analysis of SVOCs/PAHs, and provide a valuable basis of comparison. Although several SVOCs/PAHs were detected in the reference wetland surface soil samples, the number of SVOCs/PAHs detected in the CCSP wetland surface soils was appreciably less than the number of individual SVOC/PAH compounds detected in subsurface soils from the wetland portion of the site. The mean and maximum concentrations of the SVOCs/PAHs detected in both subsurface soils from the wetland and surface soils from the CCSP

wetland generally were lower for the CCSP soils, with the exception of benzaldehyde. For benzaldehyde, both the mean and maximum detected concentrations were higher for CCSP wetland surface soil samples compared to subsurface soil samples from the site wetland. SVOCs/PAHs were not detected in surface soil samples from the CCSP wetland at concentrations exceeding relevant PRGs.

There are also specific background values for certain SVOC/PAH compounds in soils at Alameda Point. Background values exist for benzo(a)pyrene, benzo(g,h,i)perylene, chrysene, fluoranthene, indeno(1,2,3-cd)pyrene, phenanthrene, and pyrene. The maximum detected concentrations of these compounds in subsurface soils from the wetland portion of the site are well below the background level developed.

SVOC/PAH data for subsurface soil samples from the site are presented on Figures E-7 and E-8 in Appendix E. Bubble plots showing the concentrations of SVOCs/PAHs in subsurface soil samples collected from the site during the RI are presented as Figures F-162 through F-234 in Appendix F-10. Bubble plots were generated for both the 1 to 4 ft bgs interval and the interval below 4 ft bgs. The description of the nature and extent of SVOCs/PAHs in wetland area subsurface soils above integrates the entire subsurface zone below 1 ft bgs. The box and whisker plots presented as Figures H-24 through H-66 in Appendix H-2 demonstrate the relationships between data for various SVOCs/PAHs in wetland area subsurface soils and CCSP reference wetland soils. These box and whisker plots also demonstrate the relationships between SVOC/PAH data in wetland area subsurface soils, wetland area surface soils, and landfill area subsurface soils.

PCBs

Aroclors 1016, 1221, 1232, and 1242 were not detected in any of the wetland area subsurface soil samples that were analyzed for PCBs. Aroclors 1248 and 1260 were both detected in approximately 13% of the wetland area subsurface soil samples, and Aroclor 1254 was detected in roughly 63% of the samples. These frequencies of detection are generally highly consistent with wetland area surface soils. The maximum detected concentrations of Aroclors 1248, 1254, and 1260 were all below 0.5 mg/kg (i.e., 0.4 mg/kg for Aroclor 1248, 0.3 mg/kg for Aroclor 1254, and 0.2 mg/kg for Aroclor 1260).

Subsurface soil sampling locations with low detected concentrations of PCB Aroclors were observed across the wetland area. The detected concentration of Aroclor 1254 marginally exceeded the Region 9 PRG at one subsurface soil sampling location in the wetland immediately adjacent to the central portion of the landfill. Overall, there is no observed pattern in the occurrence of PCBs in wetland area subsurface soils that would suggest the presence of a localized hot spot or contamination source area. Generally, the highest PCB Aroclor concentrations in subsurface soils in the wetland appear to correlate to sampling locations close to the landfill portion of the site.

Total PCB concentrations were calculable for all subsurface soil samples in the wetland portion of the site using the NOAA method. All total PCB concentrations derived according to this method were below 0.5 mg/kg, with a maximum of approximately 0.49 mg/kg. NOAA-based total PCB concentrations in wetland subsurface soils exceeded the Region 9 PRG for a limited number of sampling locations. As with individual PCB Aroclors, the highest total PCB concentrations calculated in subsurface soils for the wetland using the NOAA method appear to be located nearer the landfill portion of the site. Using the sum of Aroclors, total PCBs were calculated for 75% of the wetland area subsurface soil samples, with a maximum concentration of approximately 0.6 mg/kg. Total PCB concentrations derived from the sum of Aroclors in wetland subsurface soils exceed the

Region 9 PRG for the same locations exhibiting Region 9 PRG exceedances in the NOAA-based total PCB dataset.

Mean and maximum concentrations of individual PCB Aroclors and total PCBs detected in wetland area subsurface soils generally were highly consistent with the mean and maximum concentrations of PCB Aroclors and total PCBs detected in wetland area surface soils. Mean and maximum concentrations of individual PCB Aroclors and total PCBs detected in wetland area subsurface soils generally were significantly lower compared to mean and maximum concentrations of PCB Aroclors and total PCBs detected in landfill area subsurface soils.

Specific background values have not been developed for PCB Aroclors or total PCBs in soils at Alameda Point. In addition, subsurface soil samples were not collected from the CCSP wetland as part of the reference sampling program during the RI. However, surface soil samples were collected from the CCSP wetland for the analysis of PCBs, and provide a valuable basis for comparison. No PCB Aroclors were detected in the CCSP wetland reference soil samples. Total PCB concentrations calculated for the CCSP reference wetland samples (using the NOAA method only, as no Aroclors were detected) generally were far lower than total PCB concentrations calculated for wetland subsurface soils at IR Site 2.

Aroclor and total PCB data for subsurface soil samples from the site are presented on Figures E-9 and E-10 in Appendix E. Bubble plots showing the concentrations of PCB Aroclors and total PCBs in subsurface soil samples collected from the site during the RI are presented as Figures F-235a through F-250c in Appendix F-11. Bubble plots were generated for both the 1 to 4 ft bgs interval and the >4 ft bgs interval. The description of the nature and extent of PCBs in wetland subsurface soils above integrates the entire subsurface zone below 1 ft bgs. The box and whisker plots presented as Figures H-67a through H-74c in Appendix H-3 demonstrate the relationships between data for various Aroclors and total PCBs in wetland area subsurface soils and CCSP reference wetland surface samples. These box and whisker plots also demonstrate the relationships between Aroclor and total PCB data in wetland area subsurface soils, wetland area surface soils, and landfill area subsurface soils.

Pesticides

alpha-BHC, aldrin, *beta*-BHC, *delta*-HCH, endosulfan I, endosulfan sulfate, endrin, endrin aldehyde, endrin ketone, heptachlor, heptachlor epoxide, and toxaphene were not detected in any of the subsurface soil samples collected from the landfill portion of the site that were evaluated for pesticides. The following pesticides were detected in no more than one of the subsurface soil samples from the wetland area: 2,4-DDE; *gamma*-BHC (lindane); and methoxychlor. 2,4-DDT was detected in approximately 44% of the wetland area subsurface soil samples. 2,4-DDT, 4,4-DDT, *alpha*-chlordane, *cis*-nonachlor, dieldrin, endosulfan II, *gamma*-chlordane, and *trans*-nonachlor were detected in 63% to 75% of the wetland area subsurface soil samples analyzed for pesticides. 4,4-DDD and 4,4-DDE were detected in 94% and 100% of the wetland subsurface soil samples, respectively.

Low concentrations of pesticides were detected in subsurface soils across the wetland portion of the site. In general, there does not appear to be a clear pattern in the occurrence of pesticides, or any indication of localized hot spots or source areas in the wetlands. For all pesticides detected in wetland area subsurface soils for which there exists a PRG, the maximum detected concentration at the site was well below the relevant PRG value. Generally, the highest concentrations of pesticides were detected at sampling locations nearest the landfill portion of the site.

Mean and maximum detected concentrations of individual pesticides in wetland area subsurface soils generally were highly consistent with the mean and maximum concentrations of pesticides detected in wetland area surface soils. However, mean and maximum detected concentrations of *alpha*-chlordane, *cis*-nonachlor, *gamma*-chlordane, and *trans*-nonachlor generally were appreciably higher in wetland area subsurface soils compared to wetland area surface soils. Mean and maximum detected concentrations of individual pesticides were, however, generally far lower in wetland area subsurface soil samples compared to landfill area subsurface soil samples.

There are no specific background values for pesticides in soils at Alameda Point. In addition, reference subsurface soil samples were not collected from the CCSP wetland for the analysis of pesticides. However, surface soil samples were collected from the CCSP wetland for analysis of pesticides, and provide a valuable basis for comparison. Several pesticides were detected in the reference wetland surface soil samples. The number of pesticides detected in the CCSP wetland surface soils was slightly less than the number of individual pesticides detected in subsurface soils from the wetland portion of the site. The maximum concentrations of those pesticides detected in both surface soils from the CCSP wetland and subsurface soils from the wetland portion of the site generally were highly consistent. However, for *alpha*-chlordane, *gamma*-chlordane, and *trans*-nonachlor, the maximum detected concentration in wetland area subsurface soils was far higher than the maximum detected in CCSP reference wetland surface soils. The maximum detected concentration of 4,4-DDE was actually appreciably higher for CCSP reference wetland surface soils compared to site wetland subsurface soils.

Pesticide data for subsurface soil samples from the site are presented on Figure E-9 and E-10 in Appendix E. Bubble plots showing the concentrations of pesticides in subsurface soil samples collected from the site during the RI are presented as Figures F-251 through F-294 in Appendix F-12. Bubble plots were generated for both the 1 to 4 ft bgs interval and the interval below 4 ft bgs. The description of the nature and extent of pesticides in wetland subsurface soils above integrates the entire subsurface beneath 1 ft bgs. The box and whisker plots presented as Figures H-75 through H-103 in Appendix H-4 demonstrate the relationships between data associated with various pesticides in wetland area subsurface soils and pesticide data associated with CCSP wetland surface soils. These box and whisker plots also demonstrate the relationships between pesticide data in wetland area subsurface soils, wetland area surface soils, and landfill area subsurface soils.

VOCs

Only the following VOCs were detected in at least one subsurface soil sample collected from the wetland portion of the site: 1,4-dichlorobenzene; acetone; benzene; chlorobenzene; isopropyl benzene; and methylene chloride. Methylene chloride was detected in approximately 70% of the wetland area subsurface soil samples analyzed for VOCs (i.e., 11 of the 16 samples), whereas acetone was detected in only two of the subsurface soil samples analyzed for VOCs. Both of these compounds exhibited very low maximum detected concentrations (i.e., 9.9 µg/kg for methylene chloride and 43 µg/kg for acetone). Both acetone and methylene chloride are common laboratory contaminants. 1,4-dichlorobenzene, benzene, chlorobenzene, and isopropyl benzene were all detected in only one of the subsurface samples collected from the wetland portion of the site. The maximum detected concentrations of these compounds were all very low (i.e., 73 µg/kg for 1,4-dichlorobenzene, 1.1 µg/kg for benzene, 4.8 µg/kg for chlorobenzene, and 1.5 µg/kg for isopropyl benzene).

Overall, the occurrence of VOCs in subsurface soils in the wetland area is highly limited and VOCs occur at only very low concentrations. There is no clear evidence of significant source areas of VOCs in soil or of localized regions of significantly elevated concentrations. The maximum concentrations of all VOCs detected in subsurface soils in the wetland portion of the site were below relevant PRGs.

The number of individual VOCs detected in the wetland area subsurface soils is generally consistent with the number of individual VOCs detected in the wetland area surface soils. However, only acetone, chlorobenzene, and methylene chloride were detected in both wetland area surface soils and subsurface soils. The mean and maximum detected concentrations of these VOCs generally were consistent between the wetland area surface and subsurface soils. The number of individual VOCs detected in wetland area subsurface soils is significantly less than the number of individual VOCs detected in landfill area subsurface soils. However, mean and maximum detected concentrations of individual VOCs detected in both landfill and wetland area subsurface soils generally were both very low, with the concentrations in the landfill subsurface soils being only slightly higher in general.

Specific background values have not been developed for VOCs in soils at Alameda Point. Reference surface soil samples were collected from the CCSP wetland for analysis of VOCs, but VOCs were not detected in these reference surface soil samples from the CCSP wetland.

VOC data for soil subsurface samples from the site are presented on Figures E-11 and E-12 in Appendix E. Bubble plots showing the concentrations of VOCs in subsurface soil samples collected from the site during the RI are presented as Figures F-295 through F-342 in Appendix F-13. Bubble plots were generated for both the 1 to 4 ft bgs interval and the interval below 4 ft bgs. The description of the nature and extent of VOCs in wetland subsurface soils above integrates the entire subsurface zone below 1 ft bgs. The box and whisker plots presented as Figures H-104 through H-127 in Appendix H-5 demonstrate the relationships between data for VOCs in wetland area subsurface soils and wetland area surface soils. These box and whisker plots also demonstrate the relationships between data for VOCs in wetland area subsurface soils and landfill area subsurface soils.

PCDD/PCDF

Several individual PCDD/PCDF congeners were detected in generally 25% to 75% of the subsurface soil samples collected from the wetland area of the site that were analyzed for these compounds. As with surface soil samples, the individual PCDD/PCDF congeners detected in discrete subsurface soil samples were summed (as described in Section 5.1) to generate a TEQ for each sampling point.

For the majority of subsurface soil sampling locations where PCDD/PCDFs were analyzed in the wetland portion of the site, the calculated TEQ does not exceed the applicable PRG for 2,3,7,8-TCDD. However, the highest calculated TEQ in wetland area subsurface soils does exceed the relevant PRG. This sampling location is in the central portion of the wetland area, but does not appear to suggest a localized hot spot or a contamination source area.

For the most part, mean and maximum detected PCDD/PCDF concentrations and calculated TEQs in wetland subsurface soils were highly consistent with wetland surface soils. With very few exceptions, mean and maximum detected PCDD/PCDF concentrations and calculated TEQs in wetland subsurface soils were considerably lower compared to landfill area subsurface soils.

Subsurface soil samples were not collected from the wetland portion of CCSP. In addition, reference surface soil samples were collected from the wetland portion of CCSP, but were not analyzed for PCDD/PCDF. However, specific sampling was completed during the RI to evaluate the potential presence and extent of PCDD/PCDF contamination at Alameda Point reference locations (i.e., locations on Alameda Point but not affected in any known way by historical industrial or waste disposal activities). Specifically, soil sampling was completed at the same Alameda Point reference locations as were assessed during the radiological survey program (see Section 5.2.3). Although the reference sampling conducted at Alameda Point for PCDD/PCDF focused on surface soils only, a comparison between the reference results and the subsurface soil data from the wetland is

nevertheless valuable. Several individual PCDD/PCDF congeners were detected in most of the surface soil samples collected from the Alameda Point reference locations. The concentrations of individual PCDD/PCDF congeners detected in the Alameda Point reference samples generally were highly consistent with those detected in subsurface soil samples collected from the wetland portion of the site. TEQs calculated for the sampling locations in the Alameda Point reference areas generally were consistent with but slightly lower than TEQs calculated for subsurface soil sampling locations in the wetland portion of the site. No calculated TEQs for reference area surface soil samples exceeded the relevant PRG for 2,3,7,8-TCDD.

TEQs calculated for subsurface soil samples from the site are presented on Figures E-9 and E-10 in Appendix E. Figures F-343 and F-344 in Appendix F-14 are bubble plots of TEQs generated for subsurface soil samples collected from the site during the RI. These bubble plots represent both the 1 to 4 ft bgs interval and the interval below 4 ft bgs. The description of the nature and extent of PCDD/PCDF in wetland area subsurface soils above integrates the entire subsurface zone below 1 ft bgs. The box and whisker plots presented as Figures H-128 through H-146 in Appendix H-6 demonstrate the relationships between data for various PCDDs/PCDFs and TEQs in wetland area subsurface soils and the reference PCDD/PCDF and TEQ data developed for Alameda Point. These box and whisker plots also demonstrate the relationships between data for various PCDDs/PCDFs and TEQs in wetland area subsurface soils, wetland area surface soils, and landfill area subsurface soils.

Petroleum Hydrocarbons

DRO and RRO were detected in approximately 70 to 80% of the subsurface soil samples collected from the wetland area that were analyzed for these compounds. GRO were detected in only one of the subsurface soil samples collected from the wetland area that were analyzed for these compounds. The maximum concentrations of DRO, GRO, and RRO detected in wetland area subsurface soil samples were 150 mg/kg, 34 mg/kg, and 370 mg/kg, respectively.

In general, the frequencies of detection and detected concentrations of petroleum hydrocarbons appear to be highly consistent between wetland area subsurface soil samples and wetland area surface soil samples. However, both the mean and maximum detected concentrations of petroleum hydrocarbons in wetland area subsurface soil samples are far lower than subsurface landfill soil samples. Overall, there does not appear to be a pattern in the occurrence of petroleum hydrocarbons in wetland area subsurface soils that would suggest localized hot spots of contamination or sources areas.

Specific background values have not been developed for TPHs in soils at Alameda Point, and PRGs do not exist for DRO, GRO, or RRO. Subsurface soil samples were not collected from the CCSP wetland as part of the reference sampling program during the RI, and TPHs were not analyzed for in the reference surface soil samples collected from the CCSP wetland.

Petroleum hydrocarbon data for subsurface soil samples from the site are presented on Figures E-9 and E-10 in Appendix E. Bubble plots showing the concentrations of petroleum hydrocarbons in subsurface soil samples collected from the site during the RI are presented as Figures F-345 through F-350 in Appendix F-14. Bubble plots were generated for both the 1 to 4 ft bgs interval and the interval below 4 ft bgs. The description of the nature and extent of petroleum hydrocarbons in wetland area subsurface soils above integrates the entire subsurface zone below 1 ft bgs. The box and whisker plots presented as Figures H-147 through H-149 in Appendix H-7 demonstrate the relationships between data for the TPHs in wetland area subsurface soils and wetland area surface

soils. These box and whisker plots also demonstrate the relationships between data for TPHs in wetland area subsurface soils and landfill area subsurface soils.

Radionuclides

Lead 210 was not detected in any of the wetland area subsurface soil samples evaluated for this analyte. U-235 was detected in 75% of the wetland area subsurface soil samples analyzed for radionuclides (i.e., 3 of the 4 samples), at a maximum concentration of 0.04 pCi/g. Gross alpha gross beta, Ra-226, Ra-228, U-234, and U-238 were detected in all of the wetland area subsurface soil samples evaluated for radionuclides. The maximum concentrations of the radionuclides detected in wetland area subsurface soils were: gross alpha (15.9 pCi/g); gross beta (32 pCi/g); Ra-226 (0.44 pCi/g); Ra-228 (0.52 pCi/g); U-234 (0.61 pCi/g); and U-238 (0.65 pCi/g). These frequencies of detection and concentrations are generally highly consistent with the wetland area surface soil samples.

There does not appear to be a trend or pattern in the occurrence of radionuclides in wetland area subsurface soils that would suggest the presence of contamination hot spots or localized sources. As discussed in Section 4.0, the number of samples collected for analysis of radionuclides was limited in the wetland area. Concentrations of Ra-226 were detected that marginally exceeded the average background value developed during the 2004 radiological survey (see Section 5.2.3). However, the maximum detected Ra-226 concentration in wetland area subsurface soils is within the range of concentrations that would be expected from background conditions.

In general, the detected concentrations of radionuclides in wetland area subsurface soils are highly consistent with detected concentrations of these compounds in wetland surface soils. The detected concentrations of radionuclides in wetland area subsurface soils are also highly consistent with concentrations in landfill area subsurface soils.

No specific background values have been developed for radionuclides in soils at Alameda Point other than the value generated by the Navy during its recent survey program (see Section 5.2.3). Reference subsurface soil samples were not collected from the CCSP wetland, and radionuclides were not analyzed for in the reference surface soil samples collected from the CCSP wetland. In addition, there are no PRGs for radionuclides.

Radionuclide data for subsurface soil samples from the site are presented on Figure E-5 and E-6 in Appendix E. Bubble plots showing the concentrations of radionuclides in subsurface soil samples collected from the site during the RI are presented as Figures F-351 through F-366 in Appendix F-15. Bubble plots were generated for both the 1 to 4 ft bgs interval and the interval below 4 ft bgs. The description of the nature and extent of radionuclides in wetland area subsurface soils above integrates the entire subsurface below 1 ft bgs. The box and whisker plots presented as Figures H-150 through H-157 in Appendix H-8 demonstrate the relationships between data for radionuclides in wetland area subsurface soils, wetland area surface soils, and landfill area subsurface soils.

General Chemistry

Several subsurface soil samples collected from the wetland portion of the site were evaluated for moisture content, TOC, and/or grain-size distribution. These analytical parameters are, in and of themselves, not indicative of any type or distribution of contamination in wetland area subsurface soils, and therefore are not discussed here in detail.

5.3.2.4   Wetland FWBZ Groundwater

During the course of the RI at IR Site 2, a total of 7 groundwater samples were collected from the FWBZ in the wetland area of the site. Samples were generated during the dry season sampling program, and were collected from the shallow portion of the FWBZ using temporary wellpoints. In addition, recent quarterly groundwater monitoring completed through the Navy's ongoing monitoring program (winter 2004) coincides very closely in time with the dry season sampling event of the RI (ITSI, 2005). Ground-water samples collected through the ongoing monitoring program come from both FWBZ and SWBZ wells located along the periphery of the site and the berm between the landfill and wetlands. The wetland area FWBZ groundwater data generated specifically through the RI and wetland area FWBZ groundwater data from the recent quarterly sampling event were combined to develop the evaluation of the nature and extent of groundwater contamination in the wetland area FWBZ at the site. The following analytical parameters were evaluated in the FWBZ groundwater samples collected from the wetland, combining the RI specific data and the quarterly monitoring data described above (as noted within specific compound classes, individual analytes evaluated during the RI and the quarterly monitoring program varied to some degree):

- All 13 groundwater samples were analyzed for metals (note that six groundwater samples were analyzed for sodium), VOCs (note that no groundwater samples were analyzed for 1,1,2-trichloro-1,2,2-trifluoroethane, seven samples were analyzed for *cis*-1,3-dichloropropene, cyclohexane, methyl acetate, methylcyclohexane, and *trans*-1,3-dichloropropene, and six samples were analyzed for tert-amyl methyl ether, TBA, and tert-butyl ethyl ether), SVOCs (including PAHs) (note that seven groundwater samples were analyzed for 1,1-biphenyl, 1,2,4,5-tetrachlorobenzene, 1,4-dioxane, acetophenone, atrazine, benzaldehyde, caprolactam, and carbazole, and six groundwater samples were analyzed for 1-methylnaphthalene, benzoic acid, and NDMA), pesticides (note that seven groundwater samples were analyzed for 2,4-DDD, 2,4-DDE, 2,4-DDT, *cis*-nonachlor, and *trans*-nonachlor, and six groundwater samples were analyzed for 2,4,5,6-tetrachloro-*m*-xylene), and PCBs

- 10 groundwater samples were analyzed for TPH;

- 8 groundwater samples were analyzed for gross alpha, gross beta, and Ra-228;

- 7 groundwater samples were analyzed for tritium;

- 4 groundwater samples were analyzed for hexavalent chromium;

- 2 groundwater samples were analyzed for Pb-210, Ra-226, U-234, U-235, and U-238; and,

- 1 groundwater sample was analyzed for explosives (note that seven groundwater samples were analyzed for 2,4-dinitrotoluene, 2,6-dinitrotoluene, and nitrobenzene).

Table 5-10 provides a general statistical summary of the groundwater data used to interpret the nature and extent of contamination in the FWBZ in the wetland portion of the site. During the RI, wetland area FWBZ groundwater samples were analyzed for total and dissolved fractions of all compounds other than VOCs and general chemistry parameters. The quarterly groundwater data for wetland area FWBZ wells are only available for the total fraction for all analytes.

Metals

Arsenic, barium, copper, hexavalent chromium, iron, lead, magnesium, manganese, nickel, sodium, and zinc were detected in all of the groundwater samples collected from the wetland area FWBZ.

Aluminum and molybdenum were detected in 80% or more of the groundwater samples. Beryllium, cadmium, chromium, cobalt, thallium, and vanadium were detected in roughly 50% to 79% of the groundwater samples collected. Antimony, mercury, selenium, and silver were detected in 46%, 15%, 23%, and 31% of the groundwater samples collected, respectively.

Aluminum, arsenic, iron, lead, magnesium, manganese, molybdenum, nickel, selenium, vanadium, and zinc exceeded the established Alameda Point background values for metals in groundwater. Detected concentrations of copper, mercury, and nickel exceeded chronic marine AWQC, whereas arsenic, iron, and manganese exceeded Region 9 PRGs. Maximum concentrations of aluminum, iron, magnesium, mercury, and selenium exceeded Alameda Point background values by one order of magnitude or more. However, of these four metals, the PRG was only exceeded for iron.

Arsenic, barium, copper, iron, lead, magnesium, manganese, nickel, and zinc also were detected in all samples collected for the dissolved fraction. The following metals were not detected in the dissolved fraction of any of the wetland area FWBZ samples: aluminum; antimony; hexavalent chromium; mercury; selenium; and vanadium. Maximum concentrations for the dissolved fraction were less than half that of the total fraction for all metals except barium, cobalt, copper, magnesium, manganese, molybdenum, nickel, and silver. Concentrations of arsenic, magnesium, manganese, molybdenum, and nickel exceeded the Alameda Point background values in the dissolved fraction. However, dissolved concentrations of aluminum, iron, lead, mercury, selenium, vanadium, and zinc did not exceed the Alameda Point background values (where the total fraction of these metals did exceed Alameda Point background values). Relevant PRGs were exceeded for only arsenic and manganese for the dissolved fraction.

Higher concentrations of arsenic, mercury, and molybdenum generally were found along the southwestern and southeastern borders of the wetland. Iron and lead generally exhibited higher concentrations in the northern portion of the wetland and along the border of the wetland and landfill. Antimony, copper, magnesium, sodium, vanadium, and zinc exhibited higher concentrations along the border between the wetland and the San Francisco Bay. Selenium, nickel, and manganese generally were identified largely in the central and southern portions of the wetland. Chromium was found generally in the southwest, northwest and east central portions of the wetland.

For the majority of metals detected in both wetland and landfill area FWBZ groundwater, mean and maximum detected concentrations were lower for the wetland area FWBZ. In some cases, mean and maximum concentrations were highly consistent between wetland and landfill area FWBZ groundwater. Both mean and maximum detected concentrations of the following metals were actually higher in the wetland area FWBZ compared to the landfill area FWBZ: magnesium; mercury; molybdenum; selenium; and sodium.

Detected concentrations of metals in wetland-area FWBZ groundwater did coincide with areas that exhibited detectable concentrations of these metals in surface and/or subsurface soils. Overall, given that detectable concentrations of metals in both wetland soil and FWBZ groundwater are widespread, concentrations detected in the wetland-area FWBZ generally were not elevated significantly, and that dissolved-phase metals concentrations generally were significantly lower than in the total (i.e., unfiltered) fraction, there do not appear to be any distinct hot spots or obvious source areas for metals in the FWBZ in the wetland area.

Metals data for FWBZ groundwater samples from the site are presented on Figure E-14 in Appendix E. Bubble plots showing the concentrations of metals in groundwater samples collected from the site are presented as Figures F-371 through F-412 in Appendix F-17. Bubble plots were generated for the total and dissolved fraction of each detected analyte. The box and whisker plots presented as Figures

H-162 through H-184 in Appendix H-10 demonstrate the relationships between data for various metals in wetland area FWBZ groundwater and metals data associated with landfill area FWBZ groundwater. These box and whisker plots also demonstrate the relationship between total and dissolved metals data.

SVOCs/PAHs

1-methylnaphthalene and benzaldehyde were the only SVOCs/PAHs detected in 100% of the FWBZ groundwater samples from the wetland area. 1,4-dioxane, 2-methylnaphthalene, anthracene, fluoranthene, isophorone, naphthalene, phenanthrene, and pyrene were detected in between 36% and 67% of groundwater samples. 4-methylphenol, acenaphthene, acenaphthylene, *bis*(2-ethylhexyl) phthalate, fluorene, and *n*-nitrosodiphenylamine were detected in 15% to 35% of groundwater samples. 2-chlorophenol, 4-chloro-3-methylphenol, benzo(a)pyrene, benzo(b)fluoranthene, benzo(k)fluoranthene, butyl benzyl phthalate, chrysene, dibenzofuran, dibenzo(a,h)anthracene, and diethyl phthalate had at least one detection but were detected in less than 15% of the FWBZ groundwater samples from the wetland. All other SVOCs/PAHs were not detected in the total fraction of the groundwater. Some SVOCs/PAHs, including 2-methylphenol, acetophenone, benzo(a)anthracene, carbazole, di-*n*-octyl phthalate, *n*-nitrosodi-*n*-propylamine, and phenol, were present in the dissolved fraction of the groundwater but not the total fraction.

In general, maximum concentrations were greater in the total fraction than the dissolved fraction for all SVOCs/PAHs with the exception of 1,4-dioxane, 2-methylphenol, 4-methylphenol, acetophenone, benzo(a)anthracene, benzo(b)fluoranthene, butyl benzyl phthalate, carbazole, chrysene, di-*n*-octyl phthalate, isophorone, *n*-nitroso-*n*-propylamine, and phenol. Other SVOCs/PAHs had concentrations in the total and dissolved fraction that were consistent.

Of the SVOCs/PAHs detected in wetland area FWBZ groundwater, only *n*-nitrosodiphenylamine (in the dissolved fraction) exhibited a maximum concentration exceeding the relevant PRG. The maximum concentration of *n*-nitrosodiphenylamine was very low (0.49 μg/L), but exceeded the very conservative PRG (0.01 μg/L). For the majority of SVOCs/PAHs, detected concentrations were spread throughout the wetland area FWBZ. However, sampling location HYP14 had the highest number of individual SVOCs/PAHs detected at higher concentrations. The maximum concentrations for nearly half of the detected SVOCs/PAHs were found in HYP14. This sampling point was located in a potential oil disposal area in the northwest corner of the wetland. Soil sampling results from this area did not exhibit highly elevated concentrations of SVOCs/PAHs. However, given the shallow depth to groundwater, soil samples were collected only from generally shallow depths.

Fewer individual SVOCs/PAHs were detected in wetland area FWBZ groundwater compared to landfill area FWBZ groundwater. Overall, for those SVOCs/PAHs that were detected in both wetland area and landfill area FWBZ groundwater, mean and maximum detected concentrations were significantly lower for the wetland area FWBZ. Given that dissolved-phase SVOC/PAH concentrations were generally significantly lower than in the total (i.e., unfiltered) fraction, that SVOC/PAH concentrations were generally low and did not exceed relevant benchmarks, and that concentrations were appreciably lower in the weltand-area FWBZ compared to the landfill area, there do not appear to be any distinct hot spots or obvious source areas for SVOCs/PAHs and these compounds do not appear to represent significant contamination in FWBZ groundwater in the wetland area.

SVOC/PAH data for FWBZ groundwater samples from the site are presented on Figure E-15 in Appendix E. Bubble plots showing the concentrations of SVOCs/PAHs in groundwater samples collected from the site are presented as Figures F-413 through F-475 in Appendix F-18. Bubble plots

were generated for the total and dissolved fraction of each detected analyte. The box and whisker plots presented as Figures H-185 through H-225 in Appendix H-11 demonstrate the relationships between data for various SVOCs/PAHs in wetland area FWBZ groundwater and SVOC/PAH data associated with landfill area FWBZ groundwater. These box and whisker plots also demonstrate the relationships between total and dissolved SVOC/PAH data for the wetland area FWBZ. AWQC and Alameda Point background values are not available for SVOCs/PAHs in groundwater.

PCBs

Only Aroclor 1254 was detected in groundwater in the FWBZ in the wetland area of IR Site 2, and only in the total fraction and not the dissolved fraction of samples, consistent with the typically strong solid matrix-sorbing tendency of PCBs. This Aroclor was detected at a very low concentration (i.e., roughly 0.2 µg/L) at sampling location HYP15 between the North and South Ponds, and at approximately 1.9 µg/L at sampling location HYP14 immediately adjacent to the border of the landfill in the western portion of the site. The Aroclor 1254 concentrations detected in wetland area FWBZ groundwater at sampling locations HYP14 and HYP15 exceed the highly conservative Region 9 PRG and chronic marine AWQC for this compound.

Using the NOAA NS&T method, total PCBs were calculable in 43% of the total fraction groundwater samples (i.e., 3 of 7 groundwater samples with congener data), and in only one of the dissolved-fraction samples. The maximum NOAA-based total PCB concentration for the total fraction was approximately 3 µg/L, whereas the maximum concentration in the dissolved fraction was only 0.2 µg/L, also consistent with the tendency of PCBs to remain sorbed to the solid matrix and not dissolve. Of the calculated concentrations of total PCBs in wetland area FWBZ groundwater using the NOAA method, 3 of 4 samples (total and dissolved) exceeded the highly conservative Region 9 PRG and chronic marine AWQC. These included the NOAA-based total PCB concentration in the total fraction at sampling locations HYP14 and HYP15, and the dissolved fraction at sampling location HYP14. Using the sum of Aroclors, total PCB concentrations were calculable at only HYP14 and HYP15 (i.e., the only sampling locations where individual Aroclors were detected). The maximum total PCB concentration using the sum of Aroclors was obviously identical to the approximately 1.9 µg/L Aroclor 1254 detected at location HYP14. Both total PCB concentrations calculated using the sum of Aroclors exceeded the highly conservative Region 9 PRG and chronic marine AWQC.

Detected concentrations of PCBs were limited to FWBZ groundwater in the north and central portions of the wetland and concentrations in the dissolved sample fraction were much lower than the total fraction. Individual Aroclor and total PCB concentrations were highest at location HYP14, which was completed to coincide with a potential disposal area (oil pit) in the northwest corner of the wetland. Although the locations where PCBs were detected in wetland area FWBZ groundwater generally correspond to nearby locations where PCBs were detected in soil (both in the wetland and in the nearby landfill), the very low concentrations of PCBs in wetland-area FWBZ groundwater and the fact that PCBs were detected only in the total fraction of groundwater samples suggest that PCBs in soils do not represent a significant source of PCBs in groundwater, and that PCBs are not a significant contaminant in wetland-area FWBZ groundwater.

No individual PCB Aroclors were detected in both wetland area and landfill area FWBZ groundwater. NOAA-based total PCB concentrations calculated for wetland area and landfill area FWBZ ground-water generally were consistent. The maximum total PCB concentration based on the sum of Aroclors was higher in landfill-area FWBZ groundwater compared to wetland-area FWBZ groundwater by nearly five times.

PCB data for FWBZ groundwater samples from the site are presented on Figure E-16 in Appendix E. Bubble plots showing the concentrations of PCBs in groundwater samples collected from the site are presented as Figures F-476 through F-480b in Appendix F-19. Bubble plots were generated for the total and dissolved fraction of each detected analyte. The box and whisker plots presented as Figures H-226 through H-229b in Appendix H-12 show the relationships between data for various PCBs in wetland area FWBZ groundwater and PCB data associated with landfill area FWBZ groundwater. These box and whisker plots also demonstrate the relationships between total and dissolved PCB data. Specific Alameda Point background values have not been developed for PCBs in groundwater.

Pesticides

Only 2,4,5,6-tetrachloro-*m*-xylene, which was only analyzed during the quarterly monitoring program (ITSI, 2005), was detected in all wetland area FWBZ samples. The following pesticides were detected in 14% to 27% of the samples: 2,4-DDD; 2,4-DDE; dieldrin; *gamma*-chlordane; and methoxychlor. 4,4-DDD, 4,4-DDE, *alpha*-chlordane, *beta*-BHC, enodsulfan II, and *gamma*-BHC were detected in less than 14% of the samples. No pesticides were detected in the dissolved fraction of any groundwater samples collected from the wetland area FWBZ.

alpha-chlordane and dieldrin were the only pesticides that exceeded Region 9 PRGs and AWQC in FWBZ groundwater samples from the wetland. Note that only one groundwater sample contained a concentration of *alpha*-chlordane above the detection limit. Also, only three wetland area FWBZ groundwater samples exhibited dieldrin above detection limits, with only one of these detections exceeding the Region 9 PRG and AWQC.

Several fewer individual pesticides were detected in wetland area FWBZ groundwater compared to landfill area FWBZ groundwater. For those pesticides that were detected in both wetland area and landfill area FWBZ groundwater, mean and maximum detected analyte concentrations generally were much lower for the wetland area. However, for dieldrin, mean and maximum detected concentrations were actually appreciably higher in wetland area FWBZ groundwater. Overall, there are no apparent hot spots of pesticides in the wetland area FWBZ, and no clear source areas of pesticide contamination. Plumes of pesticide compounds mapped for FWBZ groundwater in the landfill area extend to some limited extent into the wetland, but that data suggest the plumes originate in the landfill. Given that pesticides were not detected in the dissolved fraction of any wetland area FWBZ groundwater samples, it is apparent that pesiticides in other media do not represent a source of these compounds to groundwater and that pesticides are not a significant contaminant in wetland FWBZ groundwater.

Pesticide data for FWBZ groundwater samples from the site are presented on Figure E-16 in Appendix E. Bubble plots showing the concentrations of pesticides in groundwater samples collected from the site are presented as Figures F-481 through F-506 in Appendix F-20. Bubble plots were generated for the total and dissolved fraction of each detected analyte. The box and whisker plots presented as Figures H-230 through H-257 in Appendix H-13 demonstrate the relationships between data for various pesticides in wetland area FWBZ groundwater and pesticide data associated with landfill area FWBZ groundwater. These box and whisker plots also demonstrate the relationships between total and dissolved pesticide data. Specific Alameda Point background values have not been developed for pesticides in groundwater.

VOCs

Eighteen different VOCs were detected in groundwater samples from the wetland area FWBZ. The following VOCs were detected in less than 10% of groundwater samples from the FWBZ of the wetland: 1,2-dichlorobenzene; 2-butanone (MEK); benzene; bromoform; carbon disulfide;

isopropylbenzene; *m,p*-xylenes; MTBE; *ortho*-xylene; and toluene. 1,3-dichlorobenzene, 1,4-dichlorobenzene, chlorobenzene, chloroethane, cyclohexane, and methylcyclohexane were detected in approximately 15% of the FWBZ groundwater samples from the wetland area. Acetone, which is known to be a common laboratory contaminant, was found in 23% of the samples, and TBA was detected in 83% of the samples. The maximum TBA concentration of 8.6 µg/L was measured in monitoring well M038-A, along the boundary between the landfill and the wetland.

The low Region 9 PRGs for benzene and 1,4-dichlorobenzene (0.35 and 0.5 µg/L, respectively) were exceeded by one order of magnitude, but no other PRGs were exceeded in the wetland area FWBZ groundwater. There are no relevant AWQC for VOCs. The highest concentrations of benzene and 1,4-dichlorobenzene were both detected at location HYP14, which was located in the potential former oil disposal area in the far northwestern corner of the wetland immediately adjacent to the landfill.

For those VOCs that were detected in both landfill and wetland FWBZ groundwater, the mean and maximum concentrations are highly consistent or appreciably lower in the wetland area with the exception of carbon disulfide, which was detected at a maximum concentration of 11 µg/L in the wetland and 0.40 µg/L in the landfill. Generally, VOCs in wetland area FWBZ groundwater samples were detected in the northwest corner of the wetland, near the landfill and San Francisco Bay, and in the east central portion of the wetland near the boundary between the landfill and the wetland.

Overall, VOCs that were detected in wetland FWBZ groundwater were widespread and at low-level concentrations. As with pesticides, plumes of VOCs mapped for FWBZ groundwater in the landfill area extend to some limited extent into the wetland, and the data suggest some degree of contribution to these plumes from sampling location HYP14 in the wetland. Overall, the data do not suggest that there is significant VOC contamination in wetland area FWBZ groundwater.

VOC data for FWBZ groundwater samples from the site are presented on Figure E-17 in Appendix E. Bubble plots showing the concentrations of VOCs in groundwater samples collected from the site are presented as Figures F-507 through F-534 in Appendix F-21. The box and whisker plots presented as Figures H-258 through H-286 in Appendix H-14 demonstrate the relationships between data for various VOCs in wetland area FWBZ groundwater and VOC data associated with landfill area FWBZ groundwater. In addition, isoconcentration maps for benzene, chlorobenzene, and 1,4-dichlorobenzene detected in the FWBZ groundwater are provided in Figures 5-4, 5-5, and 5-6, respectively. Specific Alameda Point background values have not been developed for VOCs in groundwater.

Petroleum Hydrocarbons

DRO, GRO, and RRO were detected in 80%, 20%, and 60%, respectively, of the FWBZ groundwater samples collected from the wetland area. The maximum concentrations of DRO, GRO, and RRO detected in wetland area FWBZ groundwater samples were 2,900, 290, and 3,400 µg/L, respectively. There are no specific Alameda Point background values, PRGs, or AWQC for TPHs in groundwater.

In general, TPH concentrations were consistent between total and dissolved sample fractions for the wetland area FWBZ. The highest concentrations of TPHs were identified generally in the northwest corner of the wetland near the landfill border and the San Francisco Bay, and in the east central portion of the wetland along the border with the landfill. Mean and maximum detected TPH concentrations generally were lower for the wetland area FWBZ compared to the landfill area FWBZ. Soil sampling in the wetland area did not demonstrate significant levels of petroleum hydrocarbons, and although higher concentrations of petroleum hydrocarbons appear to correlate to the general location

of a potential historical oil disposal pit in the northwest corner of the wetland, there is no clear source area of TPHs apparent in the wetland area based on the RI data.

TPH data for FWBZ groundwater samples from the site are presented on Figure E-16 in Appendix E. Bubble plots showing the concentrations of petroleum hydrocarbons in groundwater samples collected from the site are presented as Figures F-536 through F-541 in Appendix F-22. Bubble plots were generated for the total and dissolved fraction of each detected analyte. The box and whisker plots presented as Figures H-287 through H-289 in Appendix H-15 demonstrate the relationships between data for various petroleum hydrocarbons in wetland area FWBZ groundwater and petroleum hydrocarbon data associated with landfill area FWBZ groundwater. These box and whisker plots also demonstrate the relationships between total and dissolved TPH data.

Radionuclides

Ra-226 was detected in both wetland area FWBZ groundwater samples that were analyzed for this compound. Gross beta, U-234, and U-238 were detected in between 50% and 88% of the FWBZ groundwater samples from the wetland. Ra-228 was found in 38% of the groundwater samples and gross alpha, Pb-210, U-235, and tritium were not found in any FWBZ groundwater samples from the wetland area of the site.

The maximum concentration of the radionuclides detected in wetland area FWBZ groundwater were: gross beta (175 pCi/L); Ra-226 (0.53 pCi/L); Ra-228 (4.97 pCi/L); U-234 (0.23 pCi/L); and U-238 (0.23 pCi/L). There are no specific Alameda Point background values, PRGs, or AWQC for radionuclides in groundwater. Ra-226, Ra-228, U-234, and U-238 were all detected at one wetland area FWBZ sampling location (HYP18) in the east central portion of the wetland. Maximum concentrations of Ra-226, U-234, and U-238 detected in the wetland area FWBZ were found at this location, as were many radionuclides at low levels in soils. Gross beta was present in all quarterly monitoring wells located along the southern and western border of the wetland. Concentrations of radionuclides detected in both the total and dissolved fractions of wetland area FWBZ samples generally were highly consistent.

For those radionuclides that were detected in both wetland area and landfill area FWBZ groundwater, mean and maximum detected analyte concentrations generally were highly consistent or lower for the wetland area. Overall, there do not appear to be any hotspots or localized sources of radionuclides in wetland-area FWBZ groundwater.

Radionuclide data for FWBZ groundwater samples from the site are presented on Figure E-14 in Appendix E. Bubble plots showing the concentrations of radionuclides in groundwater samples collected from the site are presented as Figures F-542 through F-553 in Appendix F-23. Bubble plots were generated for the total and dissolved fraction of each detected analyte. The box and whisker plots presented as Figures H-290 through H-297 in Appendix H-16 demonstrate the relationships between data for various radionuclides in wetland area FWBZ groundwater and radionuclide data associated with landfill area FWBZ groundwater. These box and whisker plots also demonstrate the relationships between total and dissolved radionuclide data.

Explosives

Explosives were not detected in the FWBZ groundwater sample that was collected in the wetland area and analyzed for explosives.

5.3.2.5 Wetland SWBZ Groundwater

Groundwater in the SWBZ was evaluated using exclusively data from the winter 2004 quarterly monitoring dataset (ITSI, 2005). A total of two groundwater samples were collected from wells in the wetland area that monitor the SWBZ. The following analytical parameters were evaluated in the SWBZ groundwater samples collected from the wetland during winter 2004 based on the quarterly monitoring data:

- Both groundwater samples were analyzed for metals, VOCs, SVOCs (including PAHs), TPHs, gross alpha, gross beta, Ra-228, tritium, and the explosives 2,4-dinitrotoluene, 2,6-dinitrotoluene, and nitrobenzene.

Table 5-10 provides a general statistical summary of the SWBZ groundwater data used to interpret the nature and extent of contamination in the SWBZ in the wetland portion of the site. The wetland area SWBZ groundwater data are available only for the total fraction for all analytes.

Metals

Aluminum, antimony, arsenic, barium, chromium, cobalt, copper, iron, lead, magnesium, manganese, molybdenum, nickel, selenium, sodium, vanadium, and zinc were detected in both of the groundwater samples collected from the wetland area SWBZ. Thallium was detected in 1 of 2 groundwater samples collected. Beryllium, cadmium, mercury, and silver were not detected in any SWBZ groundwater samples in the wetland area.

The Region 9 PRG was exceeded only for arsenic in the wetland area SWBZ. No metals were detected in wetland area SWBZ groundwater exceeding relevant AWQC. Overall, mean and maximum detected concentrations of metals in wetland area SWBZ groundwater were lower that concentrations detected in the wetland area FWBZ and the landfill area SWBZ groundwater.

Given the limited amount of data available for the wetland area SWBZ, it is difficult to identify patterns in the occurrence of metals. Some locations where more elevated concentrations of metals were identified in wetland-area SWBZ groundwater correlate to locations where metals concentrations were detected in wetland FWBZ groundwater and/or soils. However, given that metals concentrations in the SWBZ were generally significantly lower than in the FWBZ, the very limited number of individual compounds that exceed relevant benchmarks in the SWBZ, and the overall limited magnitude of metals impact in wetland SWBZ groundwater, the distribution of metals in wetland-area SWBZ groundwater does not suggest the presence of any discrete contamination hotspots or sources.

Metals data for SWBZ groundwater samples from the site are presented on Figure E-14 in Appendix E. Bubble plots showing the concentrations of metals in groundwater samples collected from the site are presented as Figures F-556 through F-574 in Appendix F-25. The box and whisker plots presented as Figures H-162 through H-184 in Appendix H-10 demonstrate the relationships between data for various metals in wetland area SWBZ groundwater and metals data associated with landfill area SWBZ groundwater. These box and whisker plots also demonstrate the relationships between data for various metals in wetland area SWBZ groundwater and FWBZ groundwater.

SVOCs/PAHs

Only two SVOCs were detected in SWBZ groundwater in the wetland area. 1-methylnaphthalene was detected in both samples and *bis*(2-ethylhexyl) phthalate was detected in one of the samples. The mean and maximum concentrations for 1-methylnaphthalene were 94.5 µg/L and 101.1 µg/L,

respectively, and for *bis*(2-ethylhexyl) phthalate were both 14 µg/L. There are no Alameda Point background values or AWQC for SVOCs/PAHs. The detected concentration of *bis*(2-ethylhexyl) phthalate in wetland area SWBZ groundwater represents a marginal exceedance of the Region 9 PRG. There is no PRG for 1-methylnaphthalene. Overall, it does not appear that there is any significant SVOC/PAH contamination or significant sources of SVOC/PAH contamination in wetland area SWBZ groundwater.

The mean and maximum concentrations of 1-methylnaphthalene detected in wetland area SWBZ groundwater were consistent with concentrations detected in wetland area FWBZ groundwater and landfill area SWBZ groundwater. The mean and maximum concentrations of *bis*(2-ethylhexyl) phthalate detected in wetland area SWBZ groundwater were very low, but higher than those detected in wetland area FWBZ groundwater and landfill area SWBZ groundwater. These data do not suggest that SVOCs/PAHs represent a significant contamination issue in the wetland-area SWBZ.

SVOC/PAH data for SWBZ groundwater samples from the site are presented on Figure E-15 in Appendix E. Bubble plots showing the concentrations of SVOCs/PAHs in groundwater samples collected from the site are presented as Figures F-575 through F-576 in Appendix F-26. The box and whisker plots presented as Figures H-185 through H-225 in Appendix H-11 demonstrate the relation-ships between data for the SVOCs detected in wetland area SWBZ groundwater and SVOC/PAH data associated with landfill area SWBZ groundwater. These box and whisker plots also demonstrate the relationships between data for various SVOCs/PAHs in wetland area SWBZ groundwater and FWBZ groundwater.

VOCs

Only two VOCs were detected in SWBZ groundwater samples collected from the wetland area of the site. Acetone, which is a common laboratory contaminant, and TBA were each detected in one groundwater sample at very low concentrations (0.8 and 1.6 µg/L, respectively). There are no Alameda Point background values or AWQC for VOCs in groundwater.

The mean and maximum concentrations of the two VOCs detected in wetland area SWBZ ground-water were lower than mean and maximum concentrations of these compounds detected in wetland area FWBZ groundwater. In addition, mean and maximum concentrations of the two VOCs detected in wetland area SWBZ groundwater were lower than mean and maximum concentrations of these compounds detected in landfill area SWBZ groundwater. Overall, there does not appear to be any VOC contamination in wetland area SWBZ groundwater and the data do not suggest that VOCs represent a contamination issue in the wetland-area SWBZ.

VOC data for SWBZ groundwater samples from the site are presented on Figure E-17 in Appendix E. Bubble plots showing the concentrations of VOCs in groundwater samples collected from the site are presented as Figures F-577 through F-582 in Appendix F-27. The box and whisker plots presented as Figures H-258 through H-286 in Appendix H-14 demonstrate the relationships between data for various VOCs in wetland area SWBZ groundwater and VOC data associated with landfill area SWBZ groundwater. These box and whisker plots also demonstrate the relationships between data for various VOCs in wetland area SWBZ groundwater and FWBZ groundwater.

Petroleum Hydrocarbons

DRO was detected in both wetland area SWBZ groundwater samples, whereas RRO was detected only in one of the groundwater samples. GRO was not detected in either of the SWBZ groundwater

samples collected from the wetland area. The maximum concentrations of DRO and RRO detected in wetland area SWBZ groundwater samples were 150 and 91 µg/L, respectively.

There are no specific Alameda Point background values, PRGs, or AWQC for TPH in groundwater. DRO and RRO concentrations detected in wetland-area SWBZ groundwater were lower than concentrations detected in wetland-area FWBZ groundwater, and also generally were lower than concentrations detected in landfill-area SWBZ groundwater.

The concentrations of petroleum hydrocarbons detected in wetland-area SWBZ groundwater are very low, and do not appear to suggest the presence of any TPH contamination in the SWBZ in the wetland area of the site.

TPH data for SWBZ groundwater samples from the site are presented on Figure E-17 in Appendix E. Bubble plots showing the concentrations of petroleum hydrocarbons in groundwater samples collected from the site are presented as Figures F-583 through F-584 in Appendix F-28. The box and whisker plots presented as Figures H-287 through H-289 in Appendix H-15 demonstrate the relationships between data for petroleum hydrocarbons in wetland area SWBZ groundwater and petroleum hydrocarbon data associated with landfill area SWBZ groundwater. These box and whisker plots also demonstrate the relationships between data for petroleum hydrocarbons in wetland area SWBZ groundwater and FWBZ groundwater.

Radionuclides

Gross beta was detected in both of the SWBZ groundwater samples from the wetland area that were analyzed for this constituent. Gross alpha, Ra-228, and tritium were not detected in the wetland area SWBZ. There are no specific background values, PRGs, or AWQC for radionuclides in groundwater.

Gross beta concentrations detected in wetland area SWBZ groundwater were lower than concentrations in wetland area FWBZ and landfill area SWBZ groundwater. Overall, radionuclide concentrations detected in wetland area SWBZ groundwater were low and do not suggest the presence of radionuclide contamination or sources of radiological contamination.

Radionuclide data for SWBZ groundwater samples from the site are presented on Figure E-14 in Appendix E. Bubble plots showing the concentrations of radionuclides in groundwater samples collected from the site are presented as Figures F-585 through F-587 in Appendix F-29. The box and whisker plots presented as Figures H-290 through H-297 in Appendix H-16 demonstrate the relationships between data for radionuclides in wetland area SWBZ groundwater and radionuclides data associated with landfill area SWBZ groundwater. These box and whisker plots also demonstrate the relationships between data for various radionuclides in wetland area SWBZ groundwater and FWBZ groundwater.

Explosives

Explosives (i.e., 2,4-dinitrotoluene, 2,6-dinitrotoluene, and nitrobenzene) were not detected in the SWBZ groundwater samples collected from the wetland area.

Overall, the relative lack of contamination in the wetland area SWBZ is likely supported, in part, by the existence of the BSU/Young Bay Mud confining unit beneath the FWBZ across much of the site.

5.3.2.6 Wetland Surface Water

During the course of the RI at IR Site 2, a total of 22 surface water samples were collected from the northern and southern wetland ponds in the wetland portion of the site. Samples were generated during both the dry and wet season sampling events. The following analytical parameters were evaluated in the surface water samples collected from the wetland area:

- All 22 surface water samples were analyzed for metals, VOCs, SVOCs (including PAHs), alkalinity, and hardness;

- 21 surface water samples were analyzed for pesticides and PCBs;

- 18 surface water samples were analyzed for PCDDs/PCDFs;

- 12 surface water samples were analyzed for TPH;

- 10 surface water samples were analyzed for sulfide; and,

- 8 surface water samples were analyzed for gross alpha, gross beta, Pb-210, Ra-226, Ra-228, U-234, U-235, and U-238.

Table 5-11 provides a general statistical summary of the surface water data generated in the wetland portion of the site during the RI.

Metals

Barium, iron, magnesium, and manganese were detected in all of the surface water samples collected from the wetland ponds during both the dry and wet seasons. Arsenic was detected in 95% of the surface water samples collected at the site (i.e., in 100% of the samples collected during the dry season and 92% of the samples collected during the wet season). The following metals were detected in between 21% and 80% of the surface water samples collected during both sampling events: aluminum; antimony; cadmium; lead; mercury; molybdenum; nickel; vanadium; and zinc. Cobalt, chromium, and hexavalent chromium were detected in fewer than 20% of all surface water samples collected from the wetland ponds during both seasons. Beryllium, copper, selenium, silver, and thallium were not detected in any of the surface water samples collected from the wetland ponds during either sampling event.

During the dry season sampling event, frequencies of detection and mean and maximum concentrations of various metals varied to some degree between the North and South Pond. Although the frequencies of detection for most metals generally were consistent in both ponds during the dry season, cobalt, mercury, and nickel were not detected in any samples in the North Pond but were detected in 25%, 100%, and 100%, respectively, of the samples from the South Pond. In addition, molybdenum was detected in approximately two-thirds of the surface water samples from the North Pond, but was not detected in any surface water samples from the South Pond. For most metals that were detected in both ponds during the dry season sampling event, the mean and maximum concentrations generally were higher, in some cases appreciably, for South Pond surface water samples. During the wet season sampling event, the opposite appears to be true. Although frequencies of detection of various metals were again generally consistent, aluminum, chromium, hexavalent chromium, and molybdenum were detected in 100%, 33%, 33%, and 50% of the surface water samples from the North Pond but not in any surface water samples from the South Pond. However, antimony was detected in 100% of the wet season samples from the South Pond and not in any samples from the North Pond. For most metals that were detected in both ponds during the wet season sampling event, the mean and maximum concentrations generally were consistent or slightly higher for North Pond surface water samples.

For the North Pond, cadmium, chromium, and nickel were not detected in any surface water samples collected during the dry season, but were detected in 83%, 33%, and 83% of the samples collected during the wet season. Vanadium, which was not detected in any North Pond surface water samples collected during the wet season, was detected in 50% of the dry season surface water samples. In general, concentrations of metals detected in the North Pond during both the dry season and wet season were consistent. For the South Pond, aluminum, cobalt, hexavalent chromium, mercury, and vanadium were detected in 50%, 25%, 67%, 100%, and 75% of the surface water samples collected during the dry season, but not in any of the surface water samples collected during the wet season. Antimony and cadmium were not detected in any surface water samples collected from the South Pond during the dry season, but were detected in 100% and 50%, respectively, of wet season samples. In general, concentrations of metals detected in the South Pond during both the dry season and wet season were higher, in some cases appreciably, during the dry season.

Metals that were detected in surface water samples from the wetland ponds that exceeded established Alameda Point background concentrations for FWBZ groundwater included aluminum, arsenic, barium, chromium, lead, magnesium, manganese, mercury, nickel, vanadium, and zinc. Only nickel was detected in the South Pond at a concentration marginally exceeding the chronic marine AWQC. Overall, there is no obvious pattern in the occurrence or distribution of metals detected in wetland surface water that would indicate localized hot spots of contamination or contaminant sources. However, it does appear that metals occur at higher concentration in South Pond surface water, and that some seasonal variation exists in metals concentrations.

Surface water samples were collected from a CCSP surface water body for analysis of metals. Although individual frequencies of detection varied, the metals detected in IR Site 2 pond water samples generally were highly consistent with the metals detected in CCSP surface water samples. Only cadmium and hexavalent chromium were detected in site surface water and not in CCSP surface water. However, the concentrations of cadmium and hexavalent chromium detected in site surface water were both very low (i.e., maximum concentration of 0.3 µg/L for cadmium and 0.023 µg/L for hexavalent chromium). Mean and maximum concentrations of the metals detected in both site and CCSP surface water generally were consistent. Maximum concentrations of barium, manganese, and magnesium were appreciably higher for IR Site 2 surface water samples. However, maximum concentrations of aluminum, iron, and lead were actually higher in CCSP surface water samples.

Metals data for surface water samples from the site are shown on Figure E-18 in Appendix E. Bubble plots showing the concentrations of metals in surface water samples collected from the wetland ponds during the RI are presented as Figures F-588 through F-643 in Appendix F-30. The box and whisker plots presented as Figures H-300 through H-316 in Appendix H-18 demonstrate the relationships between data for metals in wetland pond and CCSP surface water, in addition to relationships between total and dissolved metals data.

SVOCs/PAHs

The following SVOCs/PAHs were detected in surface water samples collected from the wetland ponds during the RI: 1,4-dioxane; 2-methylnaphthalene; 2-methylphenol; 4-methylphenol; 4-nitroaniline; acenaphthene; acenaphthylene; acetophenone; anthracene; benzaldehyde; benzo(a)-anthracene; benzo(a)pyrene; benzo(b)fluoranthene; benzo(g,h,i)perylene; benzo(k)fluoranthene; caprolactam; carbazole; chrysene; dibenz(a,h)anthracene; dibenzofuran; fluoranthene; fluorene; indeno(1,2,3-cd)pyrene; isophorone; naphthalene; phenanthrene; and pyrene. Dibenz(a,h)anthracene was detected in only one of the 22 surface water samples analyzed for SVOCs/PAHs during the RI. 2-methylnaphthalene, 2-methylphenol, 4-methylphenol, 4-nitroaniline, acenaphthene, acetophenone, benzaldehyde, benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(g,h,i)perylene,

benzo(k)fluoranthene, caprolactam, carbazole, chrysene, dibenzofuran, fluorene, indeno(1,2,3-cd)pyrene, and naphthalene were detected in 9% to 50% of the surface water sampled collected from the wetland ponds during the RI. 1,4-dioxane, anthracene, fluoranthene, phenanthrene, and pyrene were detected in 59% to 82% of the surface water samples collected form the wetland ponds. Acenaphthylene and isophorone were detected in 91% of the surface water samples collected from the wetland ponds during the RI.

During the dry season sampling event, a slightly greater number of individual SVOCs/PAHs were detected in the North Pond compared to the South Pond. For the most part, for compounds that were detected in both ponds, frequencies of detection and mean and maximum detected concentrations generally were consistent. However, for several PAHs, including benzo(a)pyrene, benzo(b)fluoranthene, benzo(g,h,i)perylene, benzo(k)fluoranthene, and indeno(1,2,3-cd)pyrene, frequencies of detection and/or mean and maximum detected concentrations were appreciably higher for South Pond surface water. For the wet season sampling event, a greater number of individual SVOCs/PAHs were again detected in the North Pond compared to the South Pond. For the most part, for compounds that were detected in both ponds, frequencies of detection and mean and maximum detected concentrations generally were consistent during the wet season sampling event.

For the North Pond and South Pond, the suites of individual SVOCs/PAHs detected during both the dry and wet season sampling events generally were consistent, as were the mean and maximum concentrations of the individual compounds detected during both sampling events. One notable exception is that several PAHs that were detected in South Pond surface water during the dry season were not detected during the wet season. Specifically, benzo(a)pyrene, benzo(b)fluoranthene, benzo(g,h,i)perylene, benzo(k)fluoranthene, and indeno(1,2,3-cd)pyrene were detected in 25% to 50% of South Pond surface water sample during the dry season at maximum concentrations between 0.12 µg/L and 0.16 µg/L, but were not detected in South Pond surface water during the wet season.

Alameda Point background concentrations for FWBZ groundwater have not been established for SVOCs/PAHs. Overall, SVOC/PAH concentrations were very low in wetland pond surface water and there is no obvious pattern in the occurrence or distribution of SVOCs/PAHs detected in wetland surface water that would indicate localized hot spots of contamination or contaminant sources. However, it does appear that some degree of seasonal variability occurs in surface water VOC detections and concentrations.

Surface water samples were collected from a CCSP surface water body for analysis of SVOCs/PAHs. A few more individual SVOCs/PAHs were detected in pond surface water from IR Site 2 compared to CCSP surface water. In general, SVOCs/PAHs detected in both IR Site 2 and CCSP surface water exhibited relatively similar frequencies of detection. However, for the most part, mean and maximum detected concentrations in IR Site 2 surface water were higher than CCSP surface water.

SVOC/PAH data for surface water samples from the site are shown on Figure E-19 in Appendix E. Bubble plots showing the concentrations of SVOCs/PAHs in surface water samples collected from the wetland ponds during the RI are presented as Figures F-644 through F-755 in Appendix F-31. The box and whisker plots presented as Figures H-317 through H-353 in Appendix H-19 demonstrate the relationships between data for SVOCs/PAHs in wetland ponds and CCSP surface water, and the relationships between total and dissolved SVOC/PAH data.

PCBs

PCB Aroclors were not detected in any of the surface water samples collected from either wetland pond during either sampling event. Accordingly, total PCBs could not be calculated at the sum of

Aroclors. However, total PCBs were calculable using the NOAA method in the total fraction only of one surface water sample collected from the South Pond during the dry season. The very low total PCB concentration at this sampling location (i.e., approximately 0.05 µg/L) marginally exceeds the highly conservative chronic marine AWQC of 0.03 µg/L.

There are no specific background values for PCB Aroclors or total PCBs in surface water at Alameda Point. Reference surface water samples were collected from CCSP surface water for analysis of PCBs, but PCB Aroclors were not detected. As such, total PCB concentrations could not be calculated as the sum of Aroclors. Total PCBs also were incalculable using the NOAA method.

PCB data for surface water samples from the wetland ponds are presented on Figure E-20 in Appendix E. Bubble plots showing the concentrations of PCB Aroclors and total PCBs in (total and dissolved) surface water samples collected from the wetland ponds during the RI are presented as Figures F-756 through F-757 in Appendix F-32. The box and whisker plot presented as Figure H-354 in Appendix H-20 demonstrates the relationships between data for PCB Aroclors and for total PCBs in wetland ponds and CCSP surface water, and the relationships between total and dissolved PCB data.

Pesticides

Only two pesticides were detected in the surface water samples collected from the wetland ponds. *gamma*-Chlordane was detected in one South Pond surface water sample during the dry season at only 0.01 µg/L (note that this was the same South Pond sampling location where a low concentration of total PCBs was detected). *gamma*-BHC (lindane) was detected at a very low level, only 0.002 µg/L, in one of the duplicate surface water samples collected at one North Pond sampling location in the wet season; the duplicate sample yielded a non-detect result for this compound.

There are no specific background values for *gamma*-chlordane or lindane in groundwater at Alameda Point. The very low concentrations of *gamma*-chlordane and lindane detected in the South Pond dry season sample and North Pond wet season sample, respectively, do not exceed the very conservative acute AWQC for these compounds. However, the concentration of *gamma*-chlordane detected in the South Pond dry season sample does exceed the extremely low chronic AWQC value for this constituent. Reference surface water samples were collected from CCSP for analysis of pesticides, but no pesticides were detected.

Pesticide data for surface water samples from the wetland ponds are shown on Figure E-20 in Appendix E. Bubble plots showing the concentrations of pesticides in surface water samples collected from the wetland ponds during the RI are presented as Figures F-758 through F-761 in Appendix F-33. The box and whisker plots presented as Figures H-355 through H-356 in Appendix H-21 demonstrate the relationships between data for pesticides in wetland ponds and CCSP surface water, and the relationships between total and dissolved pesticides data.

VOCs

Twelve individual VOCs were detected in surface water samples collected from the wetland ponds. Carbon disulfide and toluene were detected in approximately 86% of all surface water samples collected from both wetland ponds during both the dry and wet season sampling events. Acetone was detected in 73% of the surface water samples collected. 1,4-dichlorobenzene, MEK, bromomethane, chlorobenzene, chloromethane, and styrene were detected in 3 to 6 of the individual surface water samples collected from the wetland ponds through the course of the RI. Benzene, isopropyl benzene,

and methylene chloride were each detected in a single surface water sample during the RI sampling. Acetone and methylene chloride are both common laboratory contaminants.

During the dry season sampling event, frequencies of detection and mean and maximum concentrations of various VOCs varied to some degree between the North and South Ponds. Although the frequencies of detection for most VOCs generally were consistent in both ponds during the dry season, bromomethane was not detected in any samples from the North Pond but was detected in 100% of the samples from the South Pond. In addition, chloromethane was detected in approximately 17% of the surface water samples from the North Pond, but was detected in 100% of the surface water samples from the South Pond. For the VOCs that were detected in both ponds during the dry season sampling event, the mean and maximum concentrations generally were highly consistent. During the wet season sampling event, frequencies of detection and mean and maximum concentrations of various VOCs again varied to some degree between the North and South Pond. For those VOCs that were detected in both ponds during the wet season (i.e., carbon disulfide, styrene, and toluene), frequencies of detection and mean and maximum detected concentrations generally were consistent. However, 1,4-dichlorobenzene, acetone, benzene, chlorobenzene, isopropyl benzene, and methylene chloride, which were not detected in any South Pond surface water samples during the wet season, were detected in 100%, 100%, 17%, 100%, 17%, and 17%, respectively, of North Pond surface water samples during the wet season.

For the North Pond, a larger number of individual VOCs were detected during the wet season compared to the dry season. MEK, acetone, carbon disulfide, chloromethane, and toluene were detected in surface water samples from the North Pond during the dry season. 1,4-dichlorobenzene, acetone, benzene, carbon disulfide, chlorobenzene, isopropyl benzene, methylene chloride, styrene, and toluene were detected in surface water samples from the North Pond during the wet season. For VOCs that were detected in North Pond surface water samples during both seasons (i.e., acetone, carbon disulfide, and toluene), frequencies of detection and mean and maximum detected concentrations generally were very consistent. For the South Pond, a larger number of individual VOCs were detected during the dry season compared to the wet season. MEK, acetone, bromomethane, carbon disulfide, chloromethane, and toluene were detected in surface water samples from the South Pond during the dry season. Carbon disulfide, styrene, and toluene were detected in surface water samples from the South Pond during the wet season. For VOCs that were detected in South Pond surface water samples during both seasons (i.e., carbon disulfide and toluene), frequencies of detection and mean and maximum detected concentrations generally were very consistent.

Alameda Point background concentrations for FWBZ groundwater have not been established for VOCs and no AWQC exist for VOCs. Overall, VOC concentrations were very low and there is no obvious pattern in the occurrence or distribution of VOCs detected in wetland surface water that would indicate localized hot spots of contamination or contaminant sources. However, it does appear that some degree of seasonal variability occurs in surface water VOC detections and concentrations.

Surface water samples were collected from CCSP for analysis of VOCs. Several more individual VOCs were detected in wetland pond surface water from IR Site 2 compared to CCSP. Only acetone, carbon disulfide, styrene, and toluene were detected in surface water samples from CCSP. In general, these VOCs exhibited similar mean and maximum detected concentrations between IR Site 2 and CCSP surface water.

VOC data for surface water samples from the site are shown on Figure E-21 in Appendix E. Bubble plots showing the concentrations of VOCs in surface water samples collected from the wetland ponds during the RI are presented as Figures F-762 through F-785 in Appendix F-34. The box and whisker

plots presented as Figures H-357 through H-368 in Appendix H-22 demonstrate the relationships between data for VOCs in pond and CCSP surface water.

Petroleum Hydrocarbons

DRO and RRO were detected in all of the surface water samples collected from the North Pond and South Pond that were analyzed for these compounds during the RI. GRO were detected in all of the surface water samples collected from the North Pond during the dry and wet seasons, and in all of the surface water samples collected from the South Pond during the dry season. GRO were not detected in any of the surface water samples collected from the South Pond during the wet season.

During the dry season, mean and maximum concentrations of DRO and RRO were significantly higher for the South Pond compared to the North Pond. The mean concentration of GRO was generally consistent for the two ponds, but the maximum concentration was significantly greater for the South Pond. During the wet season, mean and maximum concentrations of DRO and RRO were actually appreciably higher for the North Pond compared to the South Pond. Although GRO was not detected in the South Pond during the wet season, the mean and maximum concentrations of GRO in North Pond surface water were actually consistent with concentrations observed in the South Pond during the dry season.

For the North Pond, mean and maximum concentrations of all petroleum hydrocarbons increased significantly between the dry season and wet season. For the South Pond, mean and maximum concentrations of all petroleum hydrocarbons decreased significantly between the dry season and wet season. As indicated above, GRO were detected in all of the surface water samples collected from the South Pond during the dry season, but in none of the samples collected during the wet season.

Specific background values have not been developed for TPH in groundwater at Alameda Point. In addition, no AWQC are available for petroleum hydrocarbons in surface water. Reference surface water samples were collected from CCSP and analyzed for petroleum hydrocarbons. DRO, GRO, and RRO were detected in 40%, 20%, and 40%, respectively, of the surface water samples collected from CCSP. The mean concentration of RRO and the mean and maximum GRO concentrations in surface water samples from the IR Site 2 wetland ponds generally were consistent with concentrations from samples collected from CCSP. The mean and maximum DRO concentrations and the maximum RRO concentration in IR Site 2 wetland pond surface water exceeded concentrations from the CCSP surface water samples.

Overall, TPH concentrations were very low and there is no obvious pattern in the occurrence or distribution of TPHs detected in wetland surface water that would indicate localized hot spots of contamination or contaminant sources. However, it does appear that some degree of seasonal variability occurs in surface water TPH concentrations.

Petroleum hydrocarbon data for surface water samples from the site are shown on Figure E-20 in Appendix E. Bubble plots showing the concentrations of petroleum hydrocarbons in surface water samples collected from the wetland ponds during the RI are presented as Figures F-786 through F-797 in Appendix F-35. The box and whisker plots presented as Figures H-369 through H-371 in Appendix H-23 demonstrate the relationships between data for petroleum hydrocarbons in pond and CCSP surface water, in addition to the relationships between total and dissolved TPH data.

Radionuclides

Gross alpha was not detected in any surface water samples collected from the wetland ponds. U-235 was detected in only one surface water sample. Pb-210 and Ra-228 were detected in 25% of the surface water samples collected from the wetland ponds, and Ra-226 was detected in 50% of the samples. Gross beta was detected in 75% of the surface water samples colleted during the RI, and U-234 and U-238 were each detected in 88% of the surface water samples. The maximum concentrations of the radionuclides detected in surface water samples were: gross beta (870 pCi/L); Pb-210 (5.6 pCi/L); Ra-226 (0.44 pCi/L); Ra-228 (1.53 pCi/L); U-234 (1.16 pCi/L); U-235 (0.03 pCi/L); and U-238 (0.89 pCi/L).

During the dry season, gross beta, Ra-226, U-234, and U-238 were detected in surface water samples from both the North and South Pond. In general, the frequencies of detection and mean and maximum detected concentrations of these analytes were consistent between the two ponds. Ra-228 was not detected in any of the samples from the North Pond, but was detected in 100% of the samples from the South Pond. For the wet season, gross beta, U-234, and U-238 were detected in all of the surface water samples from both ponds. The mean and maximum concentrations of these analytes were highly consistent between the two ponds. Pb-210 was detected in all of the surface water samples from the North Pond, but not in any samples from the South Pond.

For the North Pond, gross beta, U-234, and U-238 were detected during both the dry and wet season. Although the frequencies of detection generally were consistent between seasons for these analytes, mean and maximum detected concentrations declined between the dry season and the wet season. Pb-210 was not detected in the North Pond surface water samples during the dry season, but was detected in 100% of the wet season samples. Conversely, Ra-226 was detected in 100% of the dry season surface water samples but not in any wet season samples. For the South Pond, gross beta, Ra-226, Ra-228, U-234, and U-238 were detected in dry season surface water samples, whereas only gross beta, U-234, and U-238 were detected in the wet season. The frequencies of detection for gross beta, U-234, and U-238 all increased from 50% to 100% between the dry and wet seasons. As with the North Pond, mean and maximum concentrations of the various radionuclides detected in the South Pond during both the dry and wet season were lower for the wet season.

Specific background values have not been developed for radionuclides in groundwater at Alameda Point and no AWQC exist for radionuclides in surface water. Reference surface water samples were collected from CCSP and analyzed for radionuclides. Gross beta, Pb-210, U-234, and U-238 were detected in 100%, 50%, 100%, and 100%, respectively, of the surface water samples collected from CCSP. In addition, U-235, which was not detected in any surface water sample collected from the IR Site 2 wetland ponds, was detected in 50% of the samples collected from CCSP. The mean and maximum concentrations of the various radionuclides detected in CCSP surface water generally were consistent with or lower than those for the wetland pond surface water.

Overall, radionuclide concentrations were very low and there is no obvious pattern in the occurrence or distribution of radionuclides detected in wetland surface water that would indicate localized hot spots of contamination or contaminant sources. However, it does appear that some relatively minor degree of seasonal variability occurs in surface water radionuclide concentrations. This seasonal variability is potentially the result of varying seasonal inputs of surface water from San Francisco Bay, which could potentially contain naturally occurring radionuclides, and precipitation, which generally does not contain radionuclides.

Radionuclide data for surface water samples from the site are shown on Figure E-18 in Appendix E. Bubble plots showing the concentrations of radionuclides in surface water samples collected from the

wetland ponds during the RI are presented as Figures F-798 through F-823 in Appendix F-36. The box and whisker plots presented as Figures H-372 through H-378 in Appendix H-24 demonstrate the relationships between data for radionuclides in pond and CCSP surface water, in addition to those relationships between total and dissolved radionuclide data.

Alkalinity, Hardness, and Sulfide

All surface water samples collected from the wetland ponds were evaluated for alkalinity and hardness. In addition, several surface water samples were analyzed for sulfides. These analytical parameters are, in and of themselves, not indicative of any type or distribution of contamination in wetland pond surface water, and therefore are not discussed here in detail.

5.3.2.7 Wetland Sediments

During the course of the RI at IR Site 2, a total of 30 sediment samples were collected from the North and South Ponds in the wetland area. The total number of sediment samples generated during the dry and wet season was 18 and 12, respectively. The dry season sediment samples were collected from 3 intervals at 6 sampling locations (0 to 0.3, 0.3 to 1.5, and 1.5 to 3 ft bgs) and all wet season sediment samples were collected from the 0 to 0.3 ft bgs interval. The following analytical parameters were evaluated in the sediment samples collected:

- All 30 sediment samples were analyzed for metals, VOCs, SVOCs (including PAHs), pesticides, PCBs, and percent solids;

- 15 sediment samples were analyzed for TPH and hexavalent chromium;

- 12 sediment samples were analyzed for sulfide, TOC, and grain-size distribution; and,

- 3 sediment samples were analyzed for gross alpha, gross beta, Pb-210, U-234, U-235, U-238, and Ra-226.

Table 5-12 provides a general statistical summary of the sediment data generated from the North and South Ponds in the wetland portion of IR Site 2, and from CCSP during the RI.

Metals

Aluminum, arsenic, barium, chromium, cobalt, copper, iron, lead, magnesium, manganese, mercury, nickel, thallium, vanadium, and zinc were detected in all sediment samples collected from the wetland ponds. Beryllium, selenium, and silver were detected in 80% or more of the sediment samples, and antimony, cadmium, and molybdenum were detected in approximately 67%, 30%, and 13% of the sediment samples collected, respectively. Hexavalent chromium was not detected in any of the 15 sediment samples analyzed for this metal.

Alameda Point specific background levels for metals have not been established for sediments; however, comparison to the background metals concentrations for soils is nevertheless valuable. Aluminum, barium, beryllium, cadmium, chromium, cobalt, copper, iron, lead, magnesium, manganese, mercury, nickel, vanadium, and zinc were detected in wetland pond sediments above the Alameda Point background levels for metals (TtEMI, 2001). The beryllium and lead exceeds were only at North Pond sampling locations. Copper, chromium, lead, and mercury concentrations exceeded the SFEI ambient sediment levels in sediment samples from the North Pond. Selenium and silver also exceeded the SFEI ambient sediment levels, but only slightly. Arsenic concentrations exceeded Alameda Point background soil levels in only two subsurface sediment samples from the North Pond, but did not exceed the SFEI ambient sediment levels in any sediment samples.

Generally, the metals concentrations in the North Pond sediment samples appear to be appreciably higher than those in South Pond sediments. The metals concentrations are generally widespread across the North Pond and tend to decrease with increasing depth.

Sediment reference samples were collected from CCSP and analyzed for metals. Aluminum, antimony, arsenic, barium, beryllium, chromium, cobalt, copper, iron, lead, magnesium, manganese, mercury, molybdenum, nickel, selenium, silver, thallium, vanadium, and zinc were detected in CCSP reference sediments samples. In most cases, the maximum metals concentration measured in CCSP reference sediments samples was highly consistent or lower than those concentrations measured in sediments from the wetland ponds given the following exceptions. The maximum concentration of aluminum, arsenic, barium, beryllium, cobalt, iron, molybdenum, and vanadium measured in CCSP reference sediments were actually higher than those measured in wetland pond sediments.

Similar to soils in the wetland area, there is no obvious pattern in the occurrence of metals concentrations in wetland pond sediments that would suggest localized hot spots of metals contamination or clear source areas. Metals detected in sediments in the wetland ponds are generally consistent with metals detected in surface and subsurface soils collected from the wetland portion of the site.

Metals data for surface and subsurface sediment samples from the wetland ponds are presented on Figures E-22 and E-23, respectively, in Appendix E. Bubble plots showing the concentrations of metals in sediment samples collected from the wetland ponds during the RI are presented as Figures F-824 through F-884 in Appendix F-37. Bubble plots were generated for both surface (0 to 0.3 ft bgs) and subsurface (0.3 to 1.5 and 1.5 to 3 ft bgs) sediments. The description of the nature and extent of metals in sediments above integrates the entire interval from 0 to 3 ft bgs. The box and whisker plots presented as Figures H-379 through H-399 in Appendix H-25 demonstrate the relationships between data for various metals in wetland area sediments and metals data associated with CCSP wetland reference sediments. These box and whisker plots also demonstrate the relationships between data for various metals detected in North Pond surface and subsurface sediments, and South Pond surface and subsurface sediments.

SVOCs/PAHs

Several SVOCs/PAHs were detected in sediment samples collected from the wetland ponds. In North Pond sediments the following SVOCs/PAHs were detected in approximately 50% to 100% of all surface or subsurface samples: 2-methylnaphthalene, acenaphthene, acenaphthylene, acetophenone, anthracene, benzaldehyde, benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(g,h,i)perylene, benzo(k)fluoranthene, butyl benzyl phthalate, chrysene, dibenz(a,h)anthracene, di-*n*-butyl phthalate, fluoranthene, fluorene, indeno(1,2,3-cd)pyrene, naphthalene, phenanthrene, phenol, and pyrene. In the South Pond sediments the following SVOCs/PAHs were detected in approximately 50% to 100% of all surface or subsurface samples: acenaphthene, anthracene, benzaldehyde, benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(g,h,i)perylene, benzo(k)fluoranthene, butyl benzyl phthalate, chrysene, fluoranthene, indeno(1,2,3-cd)pyrene, phenanthrene, phenol, and pyrene.

In general, SVOCs/PAHs concentrations are highly consistent or lower between surface and sub-surface sediments. Concentrations of benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(g,h,i)perylene, benzo(k)fluoranthene and indeno(1,2,3-cd)pyrene are consistently higher in subsurface sediments than surface sediments. The exception to this general trend is acenaphthene,

acetophenone, and benzaldehyde, for which concentrations are higher in surface sediments as compared to subsurface sediments.

Sediment reference samples were collected from CCSP and analyzed for SVOCs/PAHs. Similar to the wetland pond sediment samples, several SVOCs/PAHs were detected in CCSP sediments. The majority of SVOCs/PAHs detected in both CCSP sediments and wetland pond sediments had consistent mean and maximum concentrations; however, there were some exceptions. The mean concentrations of 2-methylnaphthalene, acetophenone, benzaldehyde, fluorene, and naphthalene were greater in CCSP reference sediment samples as compared to wetland pond samples. The maximum concentrations of 4-methylphenol, acenaphthene, anthracene, benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(g,h,i)perylene, benzo(k)fluoranthene, *bis*(2-ethylhexyl) phthalate, butyl benzyl phthalate, chrysene, dibenz(a,h)anthracene, di-*n*-butyl phthalate, fluoranthene, indeno(1,2,3-cd)pyrene, phenanthrene, phenol, and pyrene in wetland pond sediments exceeded the maxiumum concentrations measured in reference sediment samples from CCSP.

No SFEI ambient SVOCs/PAHs sediment levels were exceeded in any samples collected from the wetland ponds. Given this fact, there is no obvious pattern in the occurrence of SVOCs/PAHs in wetland pond sediments that would suggest localized hot spots of SVOCs/PAHs contamination or clear source areas.

SVOCs/PAHs data for surface and subsurface sediment samples from the wetland ponds are presented on Figures E-24 and E-25, respectively, in Appendix E. Bubble plots showing the concentrations of SVOCs/PAHs in sediment samples collected from the wetland ponds during the RI are presented as Figures F-885 through F-964 in Appendix F-38. Bubble plots were generated for both surface (0 to 0.3 ft bgs) and subsurface (0.3 to 1.5 and 1.5 to 3 ft bgs) sediments. The description of the nature and extent of SVOCs/PAHs in sediments above integrates the entire interval from 0 to 3 ft bgs. The box and whisker plots presented as Figures H-400 through H-429 in Appendix H-26 demonstrate the relationships between data for various SVOCs/PAHs in wetland area sediments and SVOCs/PAHs data associated with CCSP wetland reference sediments. These box and whisker plots also demonstrate the relationships between data for various SVOCs/PAHs detected in North Pond surface and subsurface sediments, and South Pond surface and subsurface sediments.

PCBs

PCB Aroclors 1016, 1221, 1232, 1242, 1248, and 1260 were not detected in any sediment samples collected from the wetland ponds. The only Aroclor detected was Aroclor 1254, which was detected in 100% of the surface and subsurface sediment samples from the North Pond, approximately 90% of the surface sediment samples collected from the South Pond, and approximately 67% of the subsurface sediment samples collected from the South Pond. The mean concentrations of Aroclor 1254 detected in surface and subsurface sediments of the North Pond (71.7 and 116.0 µg/kg, respectively) were higher than the mean concentrations detected in the South Pond (58.7 and 11.3 µg/kg, respectively). However, while the maximum detected concentration in South Pond surface sediment exceeded the maximum detected in North Pond surface sediment, the difference was not great and the maximum detectected concentration in North Pond subsurface sediment exceeded that found in the South Pond. In the North Pond, the mean and maximum Aroclor 1254 concentrations were higher in subsurface sediment compared to surface sediment; and in the South Pond, the mean and maximum concentrations were higher in the surface interval. The overall maximum detected concentration of Aroclor 1254 was 163.3 µg/kg in North Pond subsurface sediment.

Using the NOAA method, total PCB concentrations were calculable for 100% of the sediment samples collected from the wetland ponds. As with Aroclor 1254, the mean and maximum NOAA-based total PCB concentrations were higher in subsurface sediment compared to surface sediment in the North Pond, and the mean and maximum NOAA-based total PCB concentrations were higher in the surface interval compared to the subsurface zone of the South Pond. In addition, the North Pond exhibited higher mean and maximum NOAA-based total PCB concentrations compared to the equivalent intevals in the South Pond. The maximum total PCB concentrations in the surface and subsurface sediments of the North Pond were 106.1 and 196.8 µg/kg, respectively, compared to maximum concentrations of 96.9 and 19.72 µg/kg in surface and subsurface sediments from the South Pond, respectively. Given that only Aroclor 1254 was detected in any IR Site 2 sediment sample, the total PCB concentrations derived as the sum of Aroclors were identical to the Aroclor 1254 results. All total PCB concentrations calculated by both the NOAA method and as the sum of Aroclors for North Pond surface and subsurface sediments exceeded the SFEI ambient sediment concentration of 14.8 µg/kg. In addition, many of the total PCB concentrations calculated by both the NOAA method and as the sum of Aroclors for South Pond surface and subsurface sediments, other than those in deeper intervals, exceeded the SFEI ambient sediment concentration. These trends are consistent with the closer proximity of the North Pond to the landfill.

Surface sediment reference samples also were collected from CCSP and analyzed for PCBs. Aroclor 1254 and 1260 were measured in the CCSP reference sediments at maximum concentrations of 17.4 µg/kg and 9.5 µg/kg, respectively. The mean and maximum concentrations of Aroclor 1254 in CCSP reference sediments are highly consistent with the mean and maximum concentrations detected in South Pond subsurface sediment. Otherwise, the concentrations of Aroclor 1254 detected at CCSP are below those measured in sediments from the wetland ponds. NOAA-based total PCBs were calculable for 100% of the CCSP reference surface sediments, with a maximum concentration of 15.3 µg/kg. Total PCBs calculated as the sum of Aroclors exhibited a maximum concentration of 17.4 µg/kg , identical to the Aroclor 1254 maximum. The total PCB results from CCSP reference sediment using both the NOAA method and as the sum of Aroclors are consistent with results from South Pond subsurface sediment. Otherwise, as with Aroclor 1254, the concentrations of total PCBs detected at CCSP are below those measured in sediments from the wetland ponds. The total PCB results for sediments from CCSP suggest an ambient impact to this medium consistent with the SFEI ambient sediment level of 14.8 µg/kg.

PCB Aroclors and total PCBs data for surface and subsurface sediment samples from the wetland ponds are presented on Figures E-26 and E-27, respectively, in Appendix E. Bubble plots showing the concentrations of PCB Aroclors and total PCBs in sediment samples collected from the wetland ponds during the RI are presented as Figures F-965 through F-970b in Appendix F-39. Bubble plots were generated for both surface (0 to 0.3 ft bgs) and subsurface (0.3 to 1.5 and 1.5 to 3 ft bgs) sediments. The description of the nature and extent of PCBs in sediments above integrates the entire interval from 0 to 3 ft bgs. The box and whisker plots presented as Figures H-430 through H-431b in Appendix H-27 demonstrate the relationships between data for various PCBs in wetland area sediments and PCBs data associated with CCSP wetland reference sediments. These box and whisker plots also demonstrate the relationships between data for various PCBs detected in North Pond surface and subsurface sediments, and South Pond surface and subsurface sediments.

Pesticides

2,4-DDD, 4,4-DDD, and 4,4-DDE were all detected in 80% or more of all sediment samples from the wetland ponds. The following pesticides were detected in 30% to 60% of all sediment samples collected from the wetland ponds: 4,4-DDT; *alpha*-chlordane; *cis*-nonachlor; dieldrin; endosulfan II; *gamma*-chlordane; heptachlor epoxide; and *trans*-nonachlor. 2,4-DDE and heptachlor epoxide were

detected in approximately 20% and 29% of all sediment samples collected from the wetland ponds. 2,4-DDT, *beta*-BHC, endosulfan sulfate, and methoxychlor were each detected only once, all of which occurred in the North Pond. All other pesticides were below detection limits in the wetland pond sediment samples.

Detections of pesticides in surface sediments appear widespread across the North and South Ponds. In general, there does not appear to be a clear pattern or distribution to the occurrence of pesticides in surface sediment to indicate a hot spot. Pesticide concentrations observed in the subsurface sediment samples indicate higher concentrations in the North Pond than in the South Pond.

Dieldrin is the only pesticide with a SFEI ambient sediment level (0.44 µg/kg), and it was marginally exceeded in the North Pond where the maximum dieldrin concentration in surface sediments was 0.63 µg/kg. Sediment reference samples were collected from CCSP and analyzed for pesticides. Several pesticides were detected in the CCSP reference sediment samples, and in almost every case mean and maximum concentrations in reference sediments were higher than wetland pond sediments. This is the case for the following pesticides: 2,4-DDD, 2,4-DDE, 4,4-DDD, 4,4-DDE, and *trans*-nonachlor. For all other pesticides that were detected in sediment reference samples and wetland pond sediments, the mean and maximum concentrations are highly consistent.

Pesticides data for surface and subsurface sediment samples from the wetland ponds are presented on Figures E-26 and E-27, respectively, in Appendix E. Bubble plots showing the concentrations of pesticides in sediment samples collected from the wetland ponds during the RI are presented as Figures F-971 through F-1007 in Appendix F-40. Bubble plots were generated for both surface (0 to 0.3 ft bgs) and subsurface (0.3 to 1.5 and 1.5 to 3 ft bgs) sediments. The description of the nature and extent of pesticides in sediments above integrates the entire interval from 0 to 3 ft bgs. The box and whisker plots presented as Figures H-432 through H-451 in Appendix H-28 demonstrate the relationships between data for various pesticides in wetland area sediments and pesticides data associated with CCSP wetland reference sediments. These box and whisker plots also demonstrate the relationships between data for various pesticides detected in North Pond surface and subsurface sediments, and South Pond surface and subsurface sediments.

VOCs

Only five VOCs were detected in sediment samples collected from the wetland ponds. Acetone and methylene chloride, both of which are well-known laboratory contaminants, was detected in approximately 63% and 57%, respectively, of all wetland sediment samples. Carbon disulfide was detected in approximately 23% of the wetland pond sediment samples analyzed for VOCs. Methyl ethyl ketone (MEK or 2-butanone) and chlorobenzene were detected in only one sediment sample from the wetland ponds at concentrations of 44 and 5.2 µg/kg, respectively.

Sediment reference samples were collected from CCSP and analyzed for VOCs. Carbon disulfide and methylene chloride were the only VOCs detected in the reference sediment samples, both at concentrations similar to or higher than those concentrations measured in wetland pond sediments. There are no SFEI ambient sediment levels for VOCs, but given the small number and low concentrations of VOCs detected at the site, there does not appear to be any VOC contamination in the IR Site 2 wetland ponds.

VOCs data for surface and subsurface sediment samples from the wetland ponds are presented on Figures E-28 and E-29, respectively, in Appendix E. Bubble plots showing the concentrations of VOCs in sediment samples collected from the wetland ponds during the RI are presented as Figures F-1008 through F-1018 in Appendix F-41. Bubble plots were generated for both surface (0 to

0.3 ft bgs) and subsurface (0.3 to 1.5 and 1.5 to 3 ft bgs) sediments. The description of the nature and extent of VOCs in sediments above integrates the entire interval from 0 to 3 ft bgs. The box and whisker plots presented as Figures H-452 through H-456 in Appendix H-29 demonstrate the relationships between data for various VOCs in wetland area sediments and VOCs data associated with CCSP wetland reference sediments. These box and whisker plots also demonstrate the relationships between data for various pesticides detected in North Pond surface and subsurface sediments, and South Pond surface and subsurface sediments.

Petroleum Hydrocarbons

DRO and RRO were detected in approximately 80% of all sediment samples collected from the wetland ponds that were analyzed for these compounds. GRO was detected only in roughly 33% of all sediment samples collected from the wetland area that were analyzed for these compounds. The maximum concentrations of DRO, GRO, and RRO detected in wetland pond sediments were 190, 19, and 820 mg/kg, respectively.

Reference sediments samples were collected from CCSP and analyzed for TPHs. DRO and RRO were detected in all CCSP sediment samples, and GRO was not detected in any. The maximum concentrations of DRO and RRO detected in CCSP reference sediment samples was 60 and 390 mg/kg, respectively. Although these concentrations are lower than those that were detected in wetland sediment ponds, there does not appear to be any significant TPH contamination in the wetland pond sediments. There are no SFEI ambient sediment levels for TPHs.

TPH data for surface and subsurface sediment samples from the wetland ponds are presented on Figures E-26 and E-27, respectively, in Appendix E. Bubble plots showing the concentrations of TPHs in sediment samples collected from the wetland ponds during the RI are presented as Figures F-1019 through F-1025 in Appendix F-42. Bubble plots were generated for both surface (0 to 0.3 ft bgs) and subsurface (0.3 to 1.5 and 1.5 to 3 ft bgs) sediments. The description of the nature and extent of TPHs in sediments above integrates the entire interval from 0 to 3 ft bgs. The box and whisker plots presented as Figures H-457 through H-459 in Appendix H-30 demonstrate the relationships between data for various TPHs in wetland area sediments and TPH data associated with CCSP wetland reference sediments. These box and whisker plots also demonstrate the relationships between data for various TPHs detected in North Pond surface and subsurface sediments, and South Pond surface and subsurface sediments.

Radionuclides

Gross alpha, gross beta, U-234, and U-238 were detected in three, U-235 was detected in two, and Pb-210 and radium 226 were detected in one of the three surface sediment samples collected from the wetland ponds and analyzed for radionuclides. The maximum concentration of the radionuclides detected in the wetland area sediment samples were: gross alpha (12.3 pCi/g); gross beta (28.9 pCi/g); Pb-210 (3.1 pCi/g); Ra-226 (0.50 pCi/g); U-234 (0.95 pCi/g); U-235 (0.03 pCi/g); and U-238 (0.88 pCi/g). Reference sediment samples were collected from CCSP and these same radionuclides with the exception of Pb-210 (i.e., gross alpha, gross beta, Ra-226, U-234, U-235, and U-238) were detected at maximum concentrations of 20.3, 26.8, 1.64, 1.42, 0.06, 1.51 pCi/G, respectively. Clearly, the radionuclide levels measure in the wetland ponds are similar to or less than those levels measured in CCSP reference sediments, indicating that no obvious radionuclide contamination exists in the wetland ponds.

Radionuclide data for surface sediment samples from the wetland ponds are presented on Figures E-22 in Appendix E. Bubble plots showing the concentrations of radionuclides in sediment

samples collected from the wetland ponds during the RI are presented as Figures F-1026 through F-1032 in Appendix F-43. The box and whisker plots presented as Figures H-460 through H-466 in Appendix H-31 demonstrate the relationships between data for various radionuclides in wetland pond sediments and radionuclides data associated with CCSP wetland reference sediments. These box and whisker plots also demonstrate the relationships between data for various radionuclides detected in North Pond and South Pond sediments.

Sulfide, TOC, Percent Solids, and Grain Size Distribution

Several wetland pond surface and subsurface sediment samples were evaluated for moisture content, TOC, grain-size distribution, and/or sulfide. These analytical parameters are, in and of themselves, not indicative of any type or distribution of contamination in wetland pond sediments, and therefore are not discussed here in detail.

5.4 Summary and Conclusions

The preliminary geophysical survey implemented at IR Site 2 revealed that cover soil in the landfill is underlain by material with a widespread and diffuse pattern of electromagnetic response indicative of historically disposed waste. The wetland portion of the site appears to be largely free of such waste. Although the geophysical survey could not refine the exact type of waste present in various locations of the site, the site likely contains numerous types of waste. Based on available historical site information, the waste types likely present at the site include general municipal waste and a number of industrial or process waste streams related to industrial military operations conducted historically at Alameda Point.

Historical information related to IR Site 2 suggests that particular waste streams were disposed of in discrete portions of the landfill, and potentially in the wetland. Although the geophysical surveying did indicate potentially significant volumes of waste in several of the areas known or presumed to have received discrete waste types (e.g., drums or oil), the survey could not conclusively determine that the particular waste types are in fact present and generally did not indicate that these discrete areas contained a greater or lesser amount of waste material compared to the overall widespread condition observed at the site. Limited exploratory trenching conducted in the landfill portion of the site confirmed the presence of waste material in the subsurface. The trenches were completed in areas likely to contain significant quantities of waste based on other information generated. A wide variety of waste and debris was encountered during the trenching process, including glass, plastic, metal (e.g., posts, sheet metal, and siding), wood, canvas, paper, concrete, rubber (e.g., tire and hose), cable, boots, Styrofoam, carpeting, and fabric. No OEW, drums, cylinders, radiological waste, or other potentially hazardous materials were identified during the trenching operations.

An extensive RI was implemented to generate data suitable to characterize the nature and extent of contamination at IR Site 2. The RI was designed to investigate portions of the site most likely to contain the most significant levels of contamination, namely the interior portions of the landfill and wetlands. In addition, where appropriate, historically generated data of suitable quality and reliability were used and reference data were collected at CCSP to augment the RI data. Surface and subsurface soil, shallow groundwater, sediment, surface water, and tissue sampling was conducted during the RI, generating a substantial database of current site characterization data. Numerous compound classes were analyzed for in samples from the various environmental media. In combination with appropriately usable historical data, the RI site characterization data were evaluated to assess the overall nature and extent of contamination at IR Site 2. A number of individual compounds (e.g., certain metals, SVOCs, and pesticides) were detected in media at IR Site 2 at concentrations very similar to or even lower than concentrations of these compounds detected in the ambient environment, suggesting the site is not impacted by these potential contaminants to a degree that warrants consideration. Alternatively, a number of other individual

compounds (e.g., certain metals, pesticides, PCBs, and PCDDs/PCDFs) were detected in media at IR Site 2 at concentrations significantly higher than concentrations of these compounds in the ambient environment, suggesting that the site is likely a source for these contaminants.

For the most part, the presence of contaminants is more widespread in the landfill compared to the wetland, and in subsurface soil compared to surface soil. A number of compounds are present in landfill soil at concentrations exceeding appropriate background and reference (ambient) values, as well as relevant benchmarks. Higher concentrations of contaminants in wetland media generally occur around the North Pond and in relatively close proximity to the landfill. In addition, while there is some evidence of variability in the nature and extent of contamination in the wetland ponds between seasons and between ponds, the wetland ponds do not appear to be substantially impacted by the presence of contamination (i.e., either sediment or surface water). Specifically, compounds detected in wetland pond media were generally detected at low concentrations that did not exceed relevant background or refrence (ambient) values and/or benchmarks.

There is evidence that certain individual compounds detected in media at IR Site 2 are present in a more localized fashion. For instance, lead was detected in soil at two locations at the site at concentrations markedly higher than all other locations. Also, certain pesticides and VOCs in groundwater appear to evidence at least some type of plume behavior, with the plumes generally existing only in the landfill area and spatially oriented in a manner consistent with documented groundwater flow directions and gradients. These plumes are generally of low concentration, but appear to emanate from potential historical disposal locations and/or from locations with detected concentrations in soil. Given the presence of a generally continuous hydrogeologic confining unit between the FWBZ and SWBZ at the site, the potential for significant downward movement of any type of contamination is highly limited, and the data, which show little contaminant impact in SWBZ groundwater, support this model. With the exception of a limited number of compounds in the various environmental media assessed at the site, there are generally not clear source areas or contaminant hot spots but rather a widespread and diffuse occurrence of contaminants in the various media.

Certain compounds evaluated in determining the nature and extent of contamination in IR Site 2 media are actually likely to represent naturally occurring materials or degradation products of naturally occurring materials. For instance, the presence of U-234, U-235, and U-238 is the result of the natural presence of these radionuclides in geologic formations, and Ra-228 in site media is likely to be related to natural processes (i.e., the decay of naturally occurring Th-232). In addition, the presence of Pb-210 in site media is likely to be related to natural processes (i.e., the decay of naturally occurring U-238). It is possible that radionuclide constituents other than Ra-226 are incidentally associated with past Navy activities (e.g., uranium could have been present at trace levels in the ore used to generate radium-based paints, or Ra-228 could be related to the decay of potentially site-related Ra-226). Overall, however, such compounds are likely not the result specifically of historical site operations.

It is possible that a more deliberate, statistical attempt to resolve localized hot spots of various contaminants could alter this determination, and such an exercise may be necessary or beneficial during the remedial design and implementation stages at IR Site 2. However, the assessment of the nature and extent of contamination at IR Site 2 was reasonable and appropriate, and does not generally suggest the presence of hot spots of significant contamination, including, for the most part, the known or presumed locations of discrete historical waste disposal. Furthermore, a presumptive remedy approach (i.e., capping) to address contamination at IR Site 2 is anticipated to be effective at mitigating site risk related to any and all compounds of concern, regardless of a more detailed analysis of microscale contamination patterns.

Sections 6.0 and 7.0 of this RI Report evaluate the potential risks associated with constituents detected in environmental media at IR Site 2 from human health and ecological perspectives, respectively. Following

those sections, Section 8.0 of this RI Report evaluates the fate and transport of contamination at IR Site 2 with particular emphasis on the conclusions of the risk assessments.

6.0 HUMAN HEALTH RISK ASSESSMENT

6.1 Objectives and Approach

This section presents the methodology and results of the human health risk assessment for IR Site 2. The risk assessment begins with a review of the laboratory analyses performed on environmental media samples collected for the RI and the results of those analyses which were used to identify constituents of potential concern (COPCs) for evaluation in the risk assessment (Section 6.2). In Section 6.3, a screening risk assessment is presented that derives an estimate of the risk that a *hypothetical* resident would incur living at the site by comparing the maximum constituent concentrations measured in each environmental media to a conservative human health benchmark that is appropriate for unrestricted (i.e., residential) land use. The Tier 1 screening risk assessment includes a separate analysis of the vapor intrusion to indoor air pathway, which is not addressed in the conservative human health benchmarks. Section 6.4 presents the baseline risk assessment (BRA) for IR Site 2, which evaluates the potential for adverse human health effects resulting from exposure to site-related contamination under reasonably likely future land-use scenarios. Included in this section is: the conceptual site model for the site which identifies potential human receptors and exposure pathways (Section 6.4.1); models, exposure assumptions, and chemical- and site-specific data used to calculate exposure and risk (Sections 6.4.2 through 6.4.5); an evaluation of health effects associated with lead at the site (Section 6.4.6); and results of the BRA (Section 6.4.7).

6.2 Constituents of Potential Concern for Risk Assessment

As discussed in Sections 4.0 and Appendix B, samples of the following environmental media were collected to support the Site 2 RI: soil and groundwater from the landfill; and soil, groundwater, surface water, and sediment from the wetland and wetland ponds. Table 6-1 lists the classes of constituents that were analyzed in each media. Note that wetland pond sediment was not considered an exposure medium for human health risk assessment; therefore, it is not listed on Table 6-1 or discussed in this section. The total number of constituents analyzed varies by media because some methods were not performed on certain media (e.g., wetland soil and wetland pond surface water were not analyzed for explosives constituents) and because the list of analytes may vary for a given analytical method depending on the media analyzed (e.g., the list of VOCs generated from Method 8260 analysis of soil differs slightly from the list of VOCs generated from Method 8260 analysis of groundwater).

6.2.1 Reported Constituents Not Considered in the Risk Assessment

Certain constituents that were measured and reported were either excluded from the risk assessment or summed prior to being evaluated in the risk assessment. These constituents are summarized below along with the rationale for their exclusion or a description of the summing process that was implemented prior to the risk assessment:

- **PCDD/PCDF:** The 17 individual dioxin and furan congeners reported by the laboratory were not individually evaluated in the risk assessment. Instead, a TEQ was calculated to represent the presence of multiple congeners, as described in Section 5.1.

- **PCBs:** Analytical data for PCBs was reported two ways: as a set of 22 PCB congeners and 7 PCB Aroclors (Aroclor 1016, Aroclor 1221, Aroclor 1232, Aroclor 1242, Aroclor 1248, Aroclor 1254, and Aroclor 1260). The Aroclor data were excluded from the risk assessment because PCB congener data were originally used to calculate a total PCB concentration using and the NOAA NS&T Method (O'Connor, 1997) as described in Section 5.1 of this RI Report. However, to address observed discrepancies between total PCB values calculated using the NOAA NS&T method and individual Aroclor data

and/or the sum of Aroclors, an evaluation of potential risk from total PCBs based on the sum of Aroclors was conducted where this method is more conservative (i.e., higher) than the NOAA NS&T method value and the results are presented in the uncertainty discussion (see Section 6.4.7.4).

- **DDE, DDD, and DDT:** For each of these compounds, the laboratory measured and reported two isomers (2,4 and 4,4). The concentration of the individual isomers was summed and a total concentration was calculated for each compound for use in the risk assessment (i.e., total DDT, DDD, and DDE). Therefore, the individual isomers were excluded from the risk assessment.

- **Miscellaneous Parameters:** TOC, GRO, DRO, and RRO are composite parameters that are comprised of a mixture of chemicals. They cannot be evaluated in the risk assessment due to a lack of toxicity and physical/chemical data; consequently, these parameters were not considered for risk assessment. For aqueous samples, water quality parameters including alkalinity, hardness, and sulfides also were excluded from the risk assessment.

- **Radiological Parameters:** Certain radiological parameters including gross alpha and gross beta were excluded from the risk assessment because IR Site 2 media were also analyzed for individual radionuclides. Individual radionuclides were evaluated for inclusion in the risk assessment.

6.2.2 Constituents of Potential Concern for Risk Assessment

After eliminating the various constituents described above, 193 constituents remained that were reviewed for the purpose of identifing COPCs for inclusion in the risk assessment. Note the total number of constituents across all media is equal to 193; however, the number of constituents analyzed varies by media for reasons that were discussed in Section 6.2.1. All chemical and radiological constituents that were detected in the sample media (i.e., soil, groundwater, surface water) were designated as COPCs for quantitative evaluation in the risk assessment. Constituents that were not detected (i.e., FOD equal to zero) were not identified as COPCs and thus were not quantitatively evaluated in the BRA. Table 6-2 summarizes the number of constituents analyzed per media, the number of constituents with a FOD greater than zero (i.e., constituents designated with a W or X), which were identified as COPCs, and the number of constituents with a FOD equal to zero (i.e., constituents designated with a Y or Z).

In addition to evaluating constituents based on FOD, constituents were classified based on the ratio of the maximum detected concentration to a calculated residential PRG; or, for constituents that were not detected, the maximum detection limit was compared to the PRG. PRGs were calculated using the algorithms and default exposure assumptions for a residential exposure scenario that are provided in the User's Guide to the U.S. EPA Region 9 PRG Table (U.S. EPA, 2004b). Toxicity data from the Region 9 PRG table also was used to calculate the PRGs for chemicals that are listed on the PRG table. The values of the calculated residential PRGs may differ slightly from the U.S. EPA Region 9 PRGs due to the use of different values for certain chemical-specific properties (e.g., Henry's law constant); however, the differences between the calculated PRGs and the published Region 9 PRGs are minor and do not change the outcome of the COPC screening process. PRG values for radionuclides were obtained directly from U.S. EPA's Soil Screening Guidance for Radionuclides [U.S. EPA, 2000a]); or, for constituents that were not detected, the maximum detection limit was compared to the PRG.

The PRG comparison was done for two reasons. First, for constituents that were detected, it provides a means to quickly identify constituents that are not likely to be risk drivers for the medium (i.e., detected

constituents that have a maximum concentration below the PRG value are not likely to be risk drivers in the BRA) and constituents that could be risk drivers for the medium (i.e., constituents that have a maximum concentration greater than the PRG value). The constituents in each media that are not likely to be risk drivers are represented with a "W" on Table 6-2; and, the constituents that could be a risk driver are indicated with an "X". Second, it identifies constituents that have a FOD of zero that could not be ruled out as potential risk drivers because the MDL in one or more samples exceeds the PRG value; these constituents are designated with a "Y" on Table 6-2. Although these constituents cannot be quantitatively evaluated in the BRA because they were not detected, they are qualitatively addressed in the uncertainty section of the risk assessment (Section 6.4.7.4). A few constituents in each media could not be classified because toxicity data are not available to calculate a PRG value.

A summary of the constituents detected in each media is provided below. Tables 6-3 through 6-6 list the individual COPCs in landfill soil, wetland soil, groundwater, and surface water, respectively. Note that total depth (TD), which is used to represent the maximum depth of landfill and wetland subsurface soil samples, corresponds to the depth of the water table or 10 ft bgs, whichever is less. As described in Appendix B, groundwater generally was encountered between 2 and 4 ft bgs in the wetland area, whereas in the landfill area the depth to groundwater was more commonly encountered at approximately 8 ft bgs.

A total of 184 constituents were analyzed in landfill surface soil. Of the 184 analyzed constituents:

- 95 constituents had a FOD greater than zero and thus were identified as COPCs. Of these, 81 have a maximum concentration that is less than the residential soil PRG and 14 have a maximum concentration that is greater than the residential soil PRG.

- 86 constituents have a FOD equal to zero and thus were not identified as COPCs. Of these, 83 have a maximum MDL that is less than the residential soil PRG, indicating that detection limits were sufficiently sensitive for these constituents; and 3 have a MDL that is *greater* than the residential PRG (4-bromophenylphenylether; 4-chlorophenyl-phenylether; and *n*-nitrosodi-*n*-propylamine).

- Three constituents do not have toxicity data; therefore, a PRG could not be calculated for classifying the COPC (bromochloromethane, magnesium, and tetrachlorodibenzofuran). Of these, bromochloromethane had a FOD equal to zero and the other compounds had a FOD greater than zero.

A total of 184 constituents were analyzed in landfill surface and subsurface soil combined. Of the 184 analyzed constituents:

- 123 constituents have a FOD greater than zero and thus were identified as COPCs. Of these, 94 have a maximum concentration that is less than the residential soil PRG and 29 have a maximum concentration that is greater than the residential soil PRG.

- 58 constituents have a FOD equal to zero and thus were not identified as COPCs. Of these, 53 have a maximum MDL that is less than the residential soil PRG, indicating that detection limits were sufficiently sensitive for these constituents; and 5 have a MDL that is *greater* than the residential PRG (4-bromophenylphenylether; 4-chlorophenylphenyl-ether; *bis*(2-chloroethoxy)methane; *bis*(2-chloroethyl)ether; and *n*-nitrosodi-*n*-propylamine).

- 3 constituents do not have toxicity data; therefore, a PRG could not be calculated for classifying the COPC (bromochloromethane, magnesium, and tetrachlorodibenzofuran).

Of these, bromochloromethane had a FOD equal to zero and the other compounds had a FOD greater than zero.

A total of 170 constituents were analyzed in wetland surface soil. Of the 170 analyzed constituents:

- 78 constituents had a FOD greater than zero and thus were identified as COPCs. Of these, 69 have a maximum concentration that is less than the residential soil PRG and 9 have a maximum concentration that is greater than the residential soil PRG.

- 89 constituents have a FOD equal to zero and thus were not identified as COPCs. Of these, 87 have a maximum MDL that is less than the residential soil PRG, indicating that detection limits were sufficiently sensitive for these constituents; and 2 have a MDL that is *greater* than the residential PRG (4-chlorophenylphenylether; and Pb-210).

- 3 constituents do not have toxicity data; therefore, a PRG could not be calculated for classifying the COPC (bromochloromethane, magnesium, and tetrachlorodibenzofuran). Of these, bromochloromethane had a FOD equal to zero and the other compounds had a FOD greater than zero.

A total of 170 constituents were analyzed in wetland surface and subsurface soil combined. Of the 170 analyzed constituents:

- 83 constituents had a FOD greater than zero and thus were identified as COPCs. Of these, 74 have a maximum concentration that is less than the residential soil PRG and 9 have a maximum concentration that is greater than the residential soil PRG.

- 84 constituents have a FOD equal to zero and thus were not identified as COPCs. Of these, 82 have a maximum MDL that is less than the residential soil PRG, indicating that detection limits were sufficiently sensitive for these constituents; and 2 have a MDL that is *greater* than the residential PRG (4-chlorophenylphenylether; and Pb-210).

- 3 constituents do not have toxicity data; therefore, a PRG could not be calculated for classifying the COPC (bromochloromethane, magnesium, and tetrachlorodibenzofuran). Of these, bromochloromethane had a FOD equal to zero and the other compounds had a FOD greater than zero.

A total of 191 constituents were analyzed in landfill groundwater. Of these constituents:

- 110 constituents had a FOD greater than zero and thus were identified as COPCs. Of these, 82 have a maximum concentration that is less than the tap water PRG and 28 have a maximum concentration that is greater than the tap water PRG;

- 77 constituents have a FOD equal to zero and thus were not identified as COPCs. Of these, 55 have a maximum MDL that is less than the tap water PRG, indicating that detection limits were sufficiently sensitive for these constituents; and 22 have a MDL that is greater than the tap water PRG. Refer to the risk assessment uncertainties discussion in Section 6.4.7.4 for a listing of these constituents where MDLs exceeded the tap water PRG.

- 4 constituents do not have toxicity data; therefore, a PRG could not be calculated for classifying the constituent (2,4,5,6-tetrachloro-*m*-xylene, bromochloromethane,

magnesium, and sodium). Of these, bromochloromethane had a FOD equal to zero and the other compounds had a FOD greater than zero.

A total of 190 constituents were analyzed in wetland groundwater. Of the 189 analyzed constituents:

- 78 constituents had a FOD greater than zero and thus were identified as COPCs. Of these, 68 have a maximum concentration that is less than the tap water PRG and 10 have a maximum concentration that is greater than the tap water PRG;

- 108 constituents have a FOD equal to zero and thus were not identified as COPCs. Of these, 84 have a maximum MDL that is less than the tap water PRG, indicating that detection limits were sufficiently sensitive for these constituents; and 24 have a MDL that is greater than the tap water PRG. Refer to the risk assessment uncertainties discussion in Section 6.4.7.4 for a listing of these constituents where MDLs exceeded the tap water PRG.

- 4 constituents do not have toxicity data; therefore, a PRG could not be calculated for classifying the constituent (2,4,5,6-tetrachloro-*m*-xylene, bromochloromethane, magnesium, and sodium). Of these, bromochloromethane had a FOD equal to zero and the other compounds had a FOD greater than zero.

A total of 168 constituents were analyzed in surface water. Of the 168 analyzed constituents:

- 61 constituents had a FOD greater than zero and thus were identified as COPCs. Of these, 53 have a maximum concentration that is less than the tap water PRG and 8 have a maximum concentration that is greater than the tap water PRG;

- 105 constituents have a FOD equal to zero and thus were not identified as COPCs. Of these, 90 have a maximum MDL that is less than the tap water PRG, indicating that detection limits were sufficiently sensitive for these constituents; and 15 have a MDL that is *greater* than the tap water PRG. Refer to the risk assessment uncertainties discussion in Section 6.4.7.4 for a listing of these constituents where MDLs exceeded the tap water PRG.

- 2 constituents do not have toxicity data; therefore, a PRG could not be calculated for classifying the constituent (bromochloromethane and magnesium). Of these, bromochloromethane had a FOD equal to zero; whereas, magnesium was detected in all surface water samples.

6.3 Tier 1 Screening Risk Assessment

A screening risk assessment was conducted to derive an estimate of the risk that a *hypothetical* resident would incur living at the site. The purpose of the screening risk assessment is to determine the need for land use control remedies. If the screening assessment indicates that the site is not suitable for residential (unrestricted) land use, the Navy will use this information to determine the need for institutional controls that would ensure that future site use is limited to the land uses evaluated in the BRA. A residential (unrestricted) land use scenario is not evaluated in the BRA because this scenario is not likely for the site because the property is likely to be transferred and used for recreational purposes as described in Section 2.12, and because the site is a former landfill. In accordance with Navy policy on risk assessment, only probable future exposure scenarios should be evaluated in the BRA; and, unrealistic exposure scenarios that are not likely to take place at the site should not be evaluted (U.S. Navy, 2001).

The Navy will provide a mechanism of assurance that necessary institutional and/or land use controls will be implemented, such that the risk assessment can be evaluated prior to the actual implementation of such controls. The institutional controls may preclude such activities as digging, occupation of buildings, and groundwater pumping if these exposure routes are not evaluated in the risk assessment. It should be noted that the RWQCB has determined that shallow groundwater in the FWBZ beneath IR Site 2 is not suitable for municipal beneficial use (e.g., drinking water source) in accordance with California State Water Resources Control Board (SWRCB) Resolution No. 88-63 and RWQCB Resolution No. 89-39; therefore, residential (unrestricted) use of shallow groundwater should be precluded by this determination, and institutional controls should not be required to preclude groundwater usage.

6.3.1 Approach

The methodology for the screening risk assessment entailed comparing the maximum constituent concentrations measured in each environmental media (soil and groundwater) to a conservative human health benchmark that is appropriate for unrestricted residential land use. This comparison was made by calculating a hazard quotient (HQ) for each noncarcinogenic chemical and a risk value for each carcinogenic constituent. Both endpoints were calculated by dividing the maximum media concentration by the appropriate screening criteria, also expressed as concentration. Hence, the screening-level hazard quotients was calculated as follows:

$$HQ = \frac{Maximum\ Media\ Concentration}{Screening\ Benchmark\ Concentration} \qquad (6-1)$$

For the purpose of calculating screening-level non-cancer risks[1] (i.e., HQs), lead was treated as a noncarcinogenic even though health effects from lead are not typically expressed as an HQ; this results in an overestimation of the non-cancer screening risks. Carcinogens were evaluated by calculating a screening-level risk estimate for each constituent, which is calculated by dividing the maximum measured concentration of the constituent by the appropriate screening benchmark for the constituent. For example, for a carcinogenic constituent that has a screening benchmark based on a risk of 1×10^{-6}, the screening-level risk for the constituent is expressed as follows:

$$Risk = \left(\frac{Maximum\ Media\ Concentration}{Screening\ Benchmark\ Concentration} \right) \times 10^{-6} \qquad (6-2)$$

The total non-cancer risk and cancer risk were calculated by summing the HQs and cancer risk estimates for individual non-carcinogens and carcinogens, respectively.

Residential PRGs for soil and groundwater were used as the benchmarks for the screening risk assessment. As stated in Section 6.2.2, residential PRGs were calculated for this purpose using the methods and assumptions in the User's Guide to the U.S. EPA Region 9 PRG Table (U.S. EPA, 2004b). And, although the calculated PRGs differ slightly from the published Region 9 PRGs, these differences are minor (in most cases, the calculated PRG is lower than the Region 9 PRG) and do not have a significant effect on the calculated screening-level risks.The PRGs are relevant benchmarks for the purpose of calculating screening-level risks because (1) the site under consideration is located in U.S. EPA Region 9; (2) toxicity data used to calculate PRGs is consistent with the latest U.S. EPA (2003a) guidance for selecting toxicity data for use in human health risk assessments at Superfund sites; and, (3) the PRGs are accepted by U.S. EPA and DTSC. The PRGs are likely to be conservative (i.e., protective) for potential

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[1] Throughout this document, the term "non-cancer risk" is used to indicate non-cancer hazard (i.e., hazard quotient [HQ] in the case of a single chemical, or hazard index [HI] in the case of multiple chemicals).

human exposure scenarios at IR Site 2 because human exposures at IR Site 2 are likely to be less than residential and industrial exposures since the property is to be used for recreational purposes in the future. For radionuclides (including Ra-226 and Ra-228), generic soil screening levels recommended by the U.S. EPA *Soil Screening Guidance for Radionuclides* (U.S. EPA, 2000a) were used in the screening risk assessment. Like the Region 9 PRGs, the radionuclide screening levels were developed for residential exposures; therefore, these criteria are considered to be conservative for use at IR Site 2.

6.3.2 Results

The estimated screening risks for the Tier 1 assessment of soil and groundwater at IR Site 2 are presented on Table 6-7. Surface water was not evaluated in the Tier 1 risk assessment because surface water PRGs are not available. Tables J-1 through J-6 in Appendix J present a summary of the calculated screening-level risks for each of the six exposure media listed on Table 6-7. Tables 6-8 through 6-13 provide a summary of constituents in each media having a maximum concentration-to-PRG ratio >1. Below is a summary of the results for each medium evaluated.

- For landfill surface soil, Ra-226 is the largest contributor to the estimated screening cancer risk; individually, this compound contributes 1.33×10^{-4} to the overall risk of 2.77×10^{-4} (Table 6-8). Other, smaller risk contributors include arsenic, PCBs, PCDD/PCDF TEQ, and several SVOC/PAH compounds. The total non-cancer risk (non-cancer hazard) of 10.5 is the result of several constituents that contribute a relatively small individual non-cancer risk. It should be noted that the total non-cancer risk is overestimated because it includes an HQ for lead.

- For the combined surface and subsurface soil intervals in the landfill area, the risk-driving constituents are largely the same as those that drive risk in the surface soil interval; however, the magnitude of the risk associated with the individual compounds is generally higher due to the occurrence of higher constituent concentrations in the subsurface interval (Table 6-9). The results indicate a higher overall risk exists for the combinced surface and subsurface soil interval (1.46×10^{-3}) as compared to the surface interval alone. Note that for Ra-226, the risk is the same in the surface and combined surface/subsurface intervals; this indicates that the maxium concentration of this constituent occurs in the surface interval. For non-cancer risks, the compounds that contribute the most to the total non-cancer risk of 547 include lead (413), PCBs (43), naphthalene (43), and antimony (22). It should be noted that the total non-cancer risk is overestimated because it includes an HQ for lead.

- For wetland surface soil, the total estimated screening risk is 7.32×10^{-5} and the total non-cancer risk is 7.58 (Table 6-10). Ra-226 is the largest contributor to the cancer risk, having an individual cancer risk of 3.94×10^{-5}. Arsenic is the second largest contributor to cancer risk, at a level of 2.51×10^{-5}; however, arsenic may be a naturally occurring compound in soils at the site. As with landfill surface soil, the non-cancer risk of 7.58 is due to multiple constituents that each contribute a relatively low individual non-cancer risk. It should be noted that the total non-cancer risk is overestimated because it includes an HQ for lead.

- For the combined surface and subsurface soil intervals in the wetland area, the total estimated screening risk is 2.19×10^{-4} and the total non-cancer risk is 7.87 (Table 6-11). Cancer risk for this interval is higher than the cancer risk for the surface interval due to the presence of higher concentrations of arsenic in subsurface soil, which contributes a risk of 1.69×10^{-4}. Ra-226 is the only other significant contributor to the cancer risk,

having an individual cancer risk of 3.94×10^{-5}, which is equal to the risk contributed by this constituent in the surface soil interval. This indicates that the maxium concentration of this constituent occurs in the surface interval. It should be noted that the total non-cancer risk is overestimated because it includes an HQ for lead.

- For groundwater in the landfill area, the estimated screening cancer risk is 1.42×10^{-3} and the total non-cancer risk is 46.1 (Table 6-12). The constituents contributing the most cancer risk include arsenic (8.77×10^{-4}), PCDD/PCDF TEQ (2.71×10^{-5}), vinyl chloride (1.65×10^{-4}), and PCBs (5.92×10^{-5}). Naphthalene (22.6) and manganese (10.1) contribute the most non-cancer risk.

- For groundwater in the wetland area, the estimated screening cancer risk is 1.50×10^{-3} and the total non-cancer risk is 17.2 (Table 6-13). The constituents contributing the most cancer risk include arsenic (7.30×10^{-4}), dieldrin (6.27×10^{-4}), and PCBs (9.28×10^{-5}); together, these three constituents account for more than 90% of the total estimated screening cancer risk. Non-cancer risk of 17.2 is due to multiple constituents that each contribute a relatively low individual non-cancer risk.

Based on this assessment, the estimated cancer risks are above U.S. EPA's acceptable risk range of 1×10^{-6} to 1×10^{-4} set in the National Contingency Plan for all exposure media that were evaluated except wetland surface soil, which has a risk that is within this range. Non-cancer risks exceed the threshold Hazard Index (sum of hazard quotients) of 1.0 for all media, ranging from a low of 7.58 for landfill surface soil to a high of 547 for combined surface and subsurface soil in the landfill area. It should be emphasized that these risk estimates are for a hypothetical residential exposure scenario; and, because the site will likely be used for recreational purposes, they do not represent accurate estimates of the risk that site receptors will incur under actual future land use scenarios. The BRA (Section 6.4) provides an evaluation of predicted risks for future land-use conditions.

Furthermore, these screening-level risks are highly conservative for a number of reasons, including: risks are calculated based on maximum measured constituent concentrations rather than a site-wide average exposure point concentration; non-cancer risks have not been segregated according to effect and mechanism, but rather, it is conservatively assumed that all non-carcinogenic constituents elicit the same effect on the same target organ; naturally occurring constituents (e.g., inorganic metals such as arsenic and iron) are included in the calculation of risk; all constituents that were detected, regardless of the FOD, and that have toxicity data were included in the risk calculation; lead was included in the calculation of non-cancer risks for soil even though lead risk is typically evaluated separately, and in the case of combined surface and subsurface soil in the landfill, lead contributes the highest non-cancer risk; and, it is assumed that groundwater at the site is usable for consumptive purposes when in fact the RWQCB has determined that shallow groundwater beneath IR Site 2 is not suitable for municipal beneficial use.

6.3.3   Residential Vapor Intrusion Screening Assessment

Because PRGs do not take into account the vapor intrusion pathway, a screening assessment of this pathway was conducted separately. Vapor intrusion is the process by which VOCs that are present in soil gas or groundwater diffuse into an aboveground structure such as a home or building. Individuals that occupy the building can be exposed to the VOCs via inhalation. The purpose for evaluating this exposure pathway in the screening risk assessment is to determine if institutional controls are necessary to preclude occupation of buildings at the site. At this time, the plans for future development of the site do not exist; consequently, it is unknown whether or not buildings such as an office, visitor center, or other structures that would be occupied on a regular basis will be constructed on the property. The vapor intrusion pathway is evaluated in this risk assessment by assuming a conservative residential scenario, for which

the results can be used to determine whether future occupation of structures on the property will cause an unacceptable risk. This pathway was not further evaluated in the BRA because the future development/reuse plans are uncertain, and a realistic exposure scenario cannot be defined.

The screening assessment for the vapor intrusion pathway was conducted by comparing measured concentrations of VOCs in soil gas and groundwater to risk-based screening levels (RBSLs) for the vapor intrusion pathway. RBSLs are concentrations of VOCs in soil gas or groundwater directly beneath a home or building that will produce an indoor-air concentration that is equivalent to the maximum acceptable risk-based concentration in indoor air for a given set of exposure assumptions. Thus, RBSLs are calculated in a two-step process. First, a safe indoor air concentration is calculated for each VOC by rearranging the following equations to solve for C_{indoor} (DTSC, 2004) and substituting appropriate exposure assumptions for the scenario:

$$Risk = \frac{C_{building} * EF * ED * URF}{ATc * 365 \text{ d/yr}} \qquad (6\text{-}3)$$

$$Hazard\ Quotient = \frac{C_{building} * EF * ED * 1/RfC}{ATn * 365 \text{ d/yr}} \qquad (6\text{-}4)$$

where: $C_{building}$ = indoor air concentration ($\mu g/m^3$)
     EF   = Exposure frequency, 350 (days/year)
     ED   = Exposure duration, 30 (years)
     AT_n = Averaging time for non-carcinogens, 30 (years) equal to exposure duration
     AT_c = Averaging time for carcinogens, 70 (years)
     URF   = Inhalation unit risk factor, chemical-specific $(\mu g/m^3)^{-1}$
     RfC   = Inhalation reference concentration, chemical-specific ($\mu g/m^3$).

For this screening analysis, conservative values corresponding to a residential setting were used for the exposure parameters in the above equations. RfCs and URFs used to calculate C_{indoor} concentrations are provided and described in Section 6.4.4.3. Once $C_{building}$ has been determined, a corresponding soil-gas RBSL ($C_{soilgas}$) and groundwater RBSL ($C_{groundwater}$) is calculated using the following relationships (DTSC, 2004):

$$\alpha = \frac{C_{building}}{C_{soil\ gas}} \qquad (6\text{-}5)$$

$$C_{soil\ gas} = C_{groundwater} * H' * 1000\frac{L}{m^3} \qquad (6\text{-}6)$$

where: α = attenuation factor (dimensionless)
     $C_{soilgas}$ = concentration in soil gas ($\mu g/m^3$)
     $C_{groundwater}$ = concentration in groundwater ($\mu g/L$)
     H'   = Henry's Law Constant.

For this screening analysis, an attenuation factor of 9×10^{-4} was used in accordance with recommendations in DTSC (2004). This factor is the recommended attenuation factor for future residential buildings. A slightly lower value of 4×10^{-4} is recommended for future commercial buildings. Table J-7a in Appendix J summarizes calculated values for $C_{building}$, $C_{soilgas}$, and $C_{groundwater}$ for all VOCs that were

measured in soil gas and groundwater at IR Site 2. Values for all chemical-specific input parameters required to calculate these concentrations are also provided on the table.

Table 6-14a and Table 6-14b summarize the results of the residential vapor intrusion screening assessment. Table 6-14a summarizes the comparison of measured concentrations of VOCs in soil gas to the calculated soil-gas RBSLs for the vapor intrusion pathway; Table 6-14b summarizes the comparison of measured concentrations of VOCs in groundwater to the calculated groundwater RBSLs for the vapor intrusion pathway. Soil gas VOC data collected on seven separate sampling events starting in summer 2003 were compiled for all 13 soil-gas monitoring probes at IR Site 2 and compared to the calculated soil-gas RBSLs. Any measured VOC that was detected at a concentration that exceeded its respective residential soil-gas RBSL was identified and tabulated in Table 6-14a. This analysis reveals that from summer 2003 through spring 2005, two VOCs, benzene and naphthalene, occurred in soil-gas at IR Site 2 at a concentration that exceeded the residential soil-gas RBSL (Table 6-14a). Figures 6-1 and 6-2 are site maps showing the soil gas sampling locations along with the benzene and naphthalene concentrations in soil gas, respectively. Concentrations of benzene and naphthalene that exceeded residential RBSLs are highlighted in yellow. Benzene exceeded its residential soil-gas RBSL (109 ppbv) in two monitoring points (MG2-02-M and MG2-02-D) located in the south-central portion of the landfill by a factor of 2.20 (spring 2005) to 4.79 (summer 2004) over the seven monitoring events. During this same time period, naphthalene was sporadically detected above its residential soil-gas RBSL (663 ppbv) in four monitoring probes in the northeast and north-central portion of the landfill (MG2-04-S, MG-2-04M, MG-2-04D, and MG2-05S) with concentrations exceeding the RBSL by a factor ranging from 1.09 to 6.88.

All groundwater VOC data collected at IR Site 2 during the RI sampling events in October 2004 and March 2005 was compiled and reviewed to identify VOCs that are present at concentrations above screening levels for the vapor intrusion pathway. Only two VOCs, benzene and vinyl chloride, were detected at a concentration that exceeds their respective residential groundwater RBSL (Table 6-14b). Benzene was detected above its RBSL by a factor of 2.2 at one location in the wetland area. In the landfill area, benzene was detected above its RBSL at eight locations, with the maximum exceedance at a factor of 16. Figure 6-3 shows the benzene concentrations in the FWBZ groundwater samples that were collected during the RI, and highlights those concentrations that exceed the applicable residential RBSL in yellow. Vinyl chloride was detected in landfill groundwater at a concentration that exceeds its RBSL at two sampling locations, with the maximum exceedance corresponding to a factor of 25. Figure 6-4 shows that both exceedances were located in the northwest portion of the interior margin, to the north and west of the landfill area. There were no concentrations of vinyl chloride detected in the wetland area.

Two other VOCs, carbon tetrachloride and hexachlorobutadiene, were not detected in groundwater but had a maximum MDL in one or more samples that exceeds the groundwater RBSL; therefore, it is not possible to state conclusively that these constituents are not present at levels above their respective residential groundwater RBSLs for the vapor intrusion pathway (Table 6-14b). Only VOCs that had a detected concentration or a MDL above their RBSL concentration are listed on Table 6-14b. Table J-8a in Appendix J provides a complete list of all VOCs that were measured in groundwater in the wet and dry season RI sampling events, along with the maximum detected concentration and the maximum MDL for each VOC.

In conclusion, this vapor intrusion screening analysis indicates that three VOCs (benzene, naphthalene, and vinyl chloride) are present in the subsurface at IR Site 2 above residential RBSLs for the vapor intrusion pathway. However, based on the conservative nature of this analysis (i.e., the RBSLs are based on residential exposure assumptions and maximum concentrations were used in the comparison) and the anticipated future land use for IR Site 2, it cannot be concluded from this analysis that VOCs in groundwater or soil gas would pose an unacceptable risk to realistic future site receptors via the vapor intrusion pathway. Therefore, a refined vapor intrusion screening analysis was conducted and is

presented in the next section (Section 6.3.4) to evaluate potential vapor intrusion risks under a more realistic future land-use scenario.

6.3.4 Refined (Commercial) Vapor Intrusion Screening Assessment

Land-use plans indicate that IR Site 2 likely will be used for recreational purposes in the future; therefore, residential and commercial structures are not anticipated. Currently, no buildings exist on site, and there are no known plans to construct any buildings on site. However, a visitor center or other support building(s) could be erected. Presumably, a site worker assigned to the site (i.e., ranger) could spend a portion of their work day indoors at the visitor center. Therefore, a refined vapor intrusion assessment was conducted to evaluate the potential risk to such a receptor.

The refined vapor intrusion screening assessment was conducted by comparing measured concentrations of VOCs in soil gas and groundwater to RBSLs for the vapor intrusion pathway that are protective under a typical commercial exposure scenario. Although commercial land use is not anticipated for this site, commercial RBSLs are more restrictive than recreational RBSLs, thus providing an added measure of conservativeness to this analysis. RBSLs for this scenario were calculated as before, except that modified exposure assumptions and a modified equation were used to calculate the safe indoor-air concentrations, as shown below:

$$\text{Risk} = \frac{C_{building} * EF * ED * InR \times ET \times CSFi}{BW \times ATc * 365 \, d/yr} \tag{6-7}$$

$$\text{Hazard Quotient} = \frac{C_{building} * EF * ED * 1/RfDi \times InR \times ET}{BW \times ATn * 365 \, d/yr} \tag{6-8}$$

where:

$C_{building}$	= Indoor air concentration ($\mu g/m^3$)
EF	= Exposure frequency, 250 (days/year)
ED	= Exposure duration, 25 (years)
ET	= Exposure time, 0.33 (8 hours/24 hours)
BW	= Body weight, 70 kg
InR	= Inhalation rate (20 m^3/day)
AT_n	= Averaging time for non-carcinogens, 25 (years), equal to exposure duration
AT_c	= Averaging time for carcinogens, 70 (years)
CSF_i	= Cancer slope factor, inhalation, chemical-specific (mg/kg-day)$^{-1}$
RfD_i	= Reference dose, inhalation, chemical-specific (mg/kg/-day).

For this analysis, exposure values corresponding to a standard commercial exposure scenario were used to calculate RBSLs, including an exposure time of 8 hours per day, exposure frequency of 250 days per year, and exposure duration of 25 years. The receptor is assumed to be an adult; therefore, a total daily inhalation rate of 20 m^3/day and a body weight of 70 kg were used. An actual on-site worker that works inside a building on site (e.g., visitor center) is not likely to spend nearly as much time indoors. To account for the fractional daily exposure time, toxicity data expressed as RfDs and CSFs rather than RfCs and URFs were used in the above equations for the safe indoor air concentration. As before, soil-gas

RBSLs ($C_{soilgas}$) and groundwater RBSLs ($C_{groundwater}$) were calculated using the following relationships (DTSC, 2004):

$$\alpha = \frac{C_{building}}{C_{soil\ gas}}$$ (6-9)

$$C_{soil\ gas} = C_{groundwater} * H' * 1000\frac{L}{m^3}$$ (6-10)

where: α = Attenuation factor (dimensionless)
      $C_{soilgas}$ = Concentration in soil gas ($\mu g/m^3$)
      $C_{groundwater}$ = Concentration in groundwater ($\mu g/L$)
      H'   = Henry's Law Constant.

For this screening analysis, an attenuation factor of 4×10^{-4} was used, which corresponds to the value recommended by DTSC (2004) for future commercial buildings. Table J-7b in Appendix J summarizes calculated values for $C_{building}$, $C_{soilgas}$, and $C_{groundwater}$ for all VOCs that were measured in soil gas and groundwater at IR Site 2.

Tables 6-15a and 6-15b summarize the results of the commercial vapor intrusion screening assessment for soil gas and groundwater, respectively. Note that only those VOCs found to be present above residential risk-based screening criteria as described in Section 6.3.3 are listed on these tables.

The residential screening analysis described in Section 6.3.3 showed that from summer 2003 through spring 2005, two VOCs, benzene and naphthalene, occurred in soil gas at IR Site 2 at concentrations that exceeded residential soil-gas RBSLs (Table 6-15a). However, the commercial screening analysis demonstrates that no single VOC is present in soil gas at a concentration above the commercial screening level (Table 6-15a). Table 6-15a also provides, for each sampling event, a summation of the ratios of the maximum concentrations (or maximum MDLs if a VOC was not detected) of each carcinogenic and non-carcinogenic VOC analyzed in soil gas to their respective RBSL. The sums of ratios are all approximately equal to or less than 1. A sum of ratios for all carcinogens corresponds to a cumulative cancer risk of 1×10^{-6} and a sum of ratios for all non-carcinogens corresponds to a cumulative non-cancer risk [i.e., hazard index] of 1 (see Table 6-15a). The calculated ratios indicate that soil-gas concentrations at Site 2 do not pose an unacceptable cumulative cancer risk or non-cancer risk to a future commercial indoor receptor.

With respect to groundwater data, two VOCs, benzene and vinyl chloride, were detected in the landfill area at IR Site 2 at concentrations that slightly exceed their commercial groundwater RBSLs (Table 6-15b). These are the same two VOCs that exceed the residential screening criteria (see Table 6-14b). However, the maximum concentration of benzene exceeded its commercial RBSL by a factor of only 1.4, and the maximum concentration of vinyl chloride in the landfill exceeded its commercial RBSL by a factor of only 2.2 (Table 6-15b). Also, whereas both VOCs were commonly present at concentrations above their residential RBSLs, benzene and vinyl chloride both exceeded their commercial RBSLs in only one location (i.e., well) at the site (see Figures 6-5 and 6-6). No VOCs were detected in wetland groundwater at a concentration above a commercial RBSL. Table 6-15b also provides a summation of the ratios of the maximum concentration (or maximum MDL if a VOC was not detected) of all carcinogenic and non-carcinogenic VOCs analyzed in groundwater to their respective commercial RBSL. The sum of ratios for carcinogens is 4.3 in the landfill area and 0.67 for the wetland groundwater, which corresponds to a cumulative cancer risk of 4.3×10^{-6} and 0.67×10^{-6}, respectively; and, the sum of ratios for all non-carcinogens is 0.11 for the landfill and 0.0182 for the wetland, which corresponds to a

cumulative non-cancer risk [i.e., hazard index] of 0.11 and 0.182, respectively. These results indicate that vapor intrusion from groundwater is not likely to pose unacceptable risks to an indoor commercial receptor at the site.

In conclusion, a refined analysis of the vapor intrusion pathway was conducted that compares measured concentrations of VOCs in soil gas and groundwater to risk-based screening levels for a commercial exposure scenario. No single VOC in soil gas was detected at a concentration above its commercial RBSL, and estimated cumulative cancer risk and non-cancer risk based on all VOCs measured in soil gas are near or below 1×10^{-6} and 1, respectively, for seven separate sampling events. Two VOCs, benzene and vinyl chloride, were each detected in groundwater at one location at the site at a concentration that slightly exceeds their commercial RBSLs. Cumulative cancer risk for a commercial receptor estimated from groundwater VOC data (i.e., taking into account all VOCs measured in groundwater) slightly exceeds 1×10^{-6} for the landfill area and is less than 1×10^{-6} for the wetland area. Cumulative non-cancer risk (i.e., hazard index) for a commercial receptor estimated from groundwater data is below 1 for both areas. The exposure assumptions used to calculate the commercial RBSLs are overly conservative for a future site worker because they will likely spend only a portion of their work day indoors rather than a full 8 hours, and they will likely be inside a building at the site less than 250 days per year for 25 years. These results indicate that vapor intrusion is not likely to pose an unacceptable risk to a future indoor commercial receptor at the site.

6.4 Baseline Human Health Risk Assessment

The purpose of the BRA is to evaluate the potential for adverse human health effects resulting from exposure to site-related contamination under reasonably likely future land-use scenarios. The focus of the BRA is to evaluate potential future scenarios because the potential for chronic exposure to site-related contamination under current conditions is minimal. The landfill at IR Site 2 ceased operation in 1978 and there are currently no human inhabitants at the site. The only regular visitors to the site under current land-use conditions are USFWS employees who monitor the least tern colony located east of IR Site 2 during breeding season and patrol the wetland portion of IR Site 2 as part of their predator management plan. Some Navy personnel and their environmental consultants infrequently visit the site to monitor the environmental conditions of the site. Site access is limited to others, including the public. The eastern and northern borders of the site are enclosed with a 6-ft-tall cyclone fence and access to the site is controlled through one padlocked gate. The western and southern perimeter of the site is bordered with large riprap and the San Francisco Bay.

The BRA involved four main activities which constituent the basic framework for all risk assessments, including:

- **Data Review and Evaluation** – The data review and evaluation process assesses all analytical data generated by the site investigations for completeness and usability in the baseline assessment and generates statistical summaries of the data by constituent and sample media. Only data of sufficient quality are accepted for use in the BRA. The results of the data review and evaluation process have been discussed previously in Section 5.0 and Appendix G of this RI Report.

- **Exposure Assessment** – The exposure assessment develops a conceptual model for the site that identifies contaminated media, potential receptors, and exposure pathways by which the receptors may come into contact with the contaminants. Models (equations) are applied to quantify the magnitude of contaminant intake/exposure for each receptor and exposure route after defining appropriate values for exposure parameters to accurately describe the receptors and their anticipated activity patterns at the site.

- **Toxicity Assessment** – The toxicity assessment involves compiling published data for each contaminant that characterizes the relationship between the magnitude of exposure and the nature and magnitude of adverse health effects that may result from each exposure.

- **Risk Characterization** – Risk characterization involves estimating the magnitude of the potential adverse health effects of contaminants and making summary judgments about the nature of the human health threat to the defined receptor populations. Adverse health effects are classified into two broad categories: noncarcinogenic and carcinogenic.

Section 6.4.1 describes the human health exposure model for the site, which includes the contamined media, potential receptors, and exposure pathways by which the receptors may come into contact with the contaminants. Sections 6.4.2 and 6.4.3 describe the models (equations) that were used to quantify contaminant intake/exposure and risk for each receptor for each exposure applicable exposure pathway. Sections 6.4.4 and 6.4.5 describe the values used for exposure, site, and chemical-specific parameters in the intake/exposure and risk models. Section 6.4.6 presents and discusses the results of the BRA, including a discussion of the major uncertainties associated with the results.

6.4.1 Conceptual Site Model for Human Health Risk Assessment

The exposure assessment is based on receptor scenarios that define the conditions of exposure to chemical contamination. An exposure pathway defines the most probable path in which a receptor may come in contact with contaminated environmental media. In order for an exposure pathway to be complete, the following four elements must be present:

- A (primary) source of contamination;

- Contamination accumulation in and/or release/transport to a location or medium (e.g., air, soil, water) where exposure can occur;

- An individual or population engaged in an activity at or near the site that results in contact with the impacted media; and,

- A route of exposure (e.g., inhalation, dermal contact, ingestion) that leads to intake of contaminants by the individuals in the exposed population.

The conceptual site model for the site is shown in Figure 6-7. The primary source of contamination is waste that was historically emplaced in the landfill at IR Site 2. The waste is comprised of a hetero-geneous mixture of waste material in a soil matrix occurring in a layer that extends from near ground surface to a depth that approximately corresponds with the groundwater table. Part of the waste layer is likely in contact with, or during particular times of the year, submerged in groundwater, and the waste is overlain with a thin, spotty layer of cover soil of approximately 4 inches to 3 ft in thickness. Contamina-tion potentially may have been transported from the original waste-soil mixture to the surrounding envi-ronment via such mechanisms as erosion of the soil cover (or waste when/where exposed at land surface), overland transport, volatilization, and/or leaching, thereby potentially resulting in the contamination of other media and portions of the site. The wetlands area of the site is believed to be outside the area where waste was emplaced. However, based on the RI sampling results and information presented in Section 5.0, contamination from the landfill portion of the site has potentially impacted this area, including the wetland soils and sediment/surface water in the wetland ponds. In addition, as described in Section 5.3.2, the southern portion of the site (including the landfill and wetland) served as a location for the deposition of dredge spoils from Oakland Inner Harbor, the pier area and turning basin of Alameda Point, and

Seaplane Lagoon. Also, there are reports that indicate that scrap metal may have been disposed of in the wetland area by Navy PWC personnel. Possible exposure media include soil, groundwater, and surface water/sediment in the ponds in the wetlands portion of the site. Receptors and mechanisms of exposure associated with each exposure medium varies somewhat, as illustrated in Figure 6-7.

Current property transfer plans indicate that IR Site 2 will likely be used for recreational purposes such as a wildlife refuge in the future. Potential receptors present at the site might be engaged in activities including guiding tours, conducting wildlife surveys, and maintaining or restoring habitat. Such activities could expose site receptors to contaminants in soil, groundwater, or surface water at the site. Portions of the site will likely be accessible to the general public for limited recreational uses such as bird watching and walking tours. It is unlikely that the general public will be allowed uncontrolled access to the site due to concerns related to potential ecological impacts associated with the wildlife refuge. Although the site will not be used for residential or commercial/industrial purposes in the future, development of the site for recreational purposes such as a wildlife refuge could include construction of a limited number of buildings (e.g., visitor center, support buildings), utilites, and roads or paths. Therefore, workers engaged in short term excavation/construction activities at the site also have the potential to be exposed to contaminated media.

Based on the anticipated future use of the site, four receptors were identified for evaluation in the BRA, including: (1) a Future Site Worker that is a tour guide; (2) a Future Site Worker that is engaged in habitat restoration activities; (3) a Site Visitor (child and adult); and (4) a Construction/Excavation Worker engaged in short-term construction activities on site. Each receptor is described below and the corresponding exposure scenarios (i.e., exposure routes and exposure media) that were evaluated for each receptor are discussed. General exposure parameters for each receptor are also described in this section, including exposure frequency, exposure duration, and body weight. Additional exposure parameters required to calculate contaminant intake are described in Section 6.4.2 and 6.4.3.

6.4.1.1 Future Site Worker – Park Ranger/Tour Guide

One of the receptors evaluated in the risk assessment is a tour guide. This receptor is based on the Tour Guide that was evaluated in the baseline human health risk assessment for the Naval Weapons Station (NWS) Seal Beach, Sites 5 and 6, as recommended by the U.S. EPA (Bechtel, 2001). The Park Ranger/Tour Guide is evaluated as an adult employee (body weight of 70 kg) that will work at the site for a period of 25 years, 250 days per year. The primary job responsibility of the Park Ranger/Tour Guide is to conduct guided walking tours of the site for visitors. In the risk assessment for NWS Seal Beach, it was stated that walking tours are expected to require 2 hours, only one of which would be spent outdoors at the site (the other hour would be spent indoors providing a presentation to Site Visitors). This is consistent with a planned future reuse of recreational, which requires that human activities result in minimal disturbance of wildlife. Therefore, this assumption was included in this risk assessment for IR Site 2. Based on this receptor's anticipated activities at the site, the following exposure pathways were evaluated in the IR Site 2 risk assessment:

- Ingestion, dermal contact, and inhalation of wind-blown dust containing adsorbed contaminants (evaluated for surface soil);

- Inhalation of VOCs in soil (evaluated for surface and subsurface soil); and,

- External radiation from the presence of radiological constituents in soil (evaluated for surface and subsurface soil).

It is unlikely that this receptor would be exposed to groundwater or surface water at the site; therefore, an assessment of exposure to these media was not conducted for the Park Ranger/Tour Guide.

6.4.1.2 Future Site Worker – Ranger/Habitat Restoration Supervisor

An individual that supervises habitat restoration activities at the site was also evaluated in the risk assessment for IR Site 2. This receptor is based on the Habitat Restoration Supervisor that was evaluated in the baseline human health risk assessment for the NWS Seal Beach, Sites 5 and 6, as recommended by the U.S. EPA (Bechtel, 2001). The Ranger/Habitat Restoration Supervisor is evaluated as an adult employee (70 kg body weight) that will work at the site for a period of 3 years, 100 days per year. The primary job responsibility of the Ranger/Habitat Restoration Supervisor is to oversee activities including removing non-native vegetation, planting native vegetation, installing a drip irrigation system, and maintaining the restored area by performing activities such as weeding, mowing, and irrigation system repair. Although volunteers under the supervision of the Ranger/Habitat Restoration Supervisor likely also would conduct the work, it was assumed that this receptor would be actively engaged in the restoration work. For the NWS Seal Beach Risk Assessment, it was estimated that the restoration work would be completed in one year, working 8 hours per week on weekends only (i.e., 4 hours per day for two days each week), and the maintenance work would be completed over a period of two years following habitat restoration by one person working 8 hours per week (also assumed to occur in 4 hours per day for two days each week). These assumptions were included in this risk assessment for IR Site 2. Based on this receptor's anticipated activities at the site, the following exposure pathways were evaluated in the IR Site 2 risk assessment:

- Inhalation of wind-blown dust containing adsorbed contaminants (evaluated for surface soil);

- Ingestion, dermal contact, and inhalation of VOCs in soil (evaluated for surface and subsurface soil); and,

- External radiation from the presence of radiological constituents in soil (evaluated for surface and subsurface soil).

Because this receptor is engaged in restoration activities that could involve digging into the subsurface soil, it was assumed that the receptor could be exposed to contaminants in both surface soil and subsurface soil, except for dust inhalation, which was evaluated for surface soil only. This is based on the assumption that the localized nature of any digging activities associated with habitat restoration would not expose large areas/volumes of subsurface soil to wind suspension.

In addition to soil exposures, it is possible that the Ranger/Habitat Restoration Supervisor also could be exposed to contaminants in surface water if it is necessary to restore habitat adjacent to the wetland ponds at the site. Furthermore, it is also possible that digging associated with restoration activities could expose this receptor to contaminants in shallow groundwater. Therefore, dermal contact with surface water and shallow groundwater were evaluated for the Ranger/Habitat Restoration Supervisor in the risk assessment for IR Site 2, even though these pathways were not included in the NWS Seal Beach risk assessment.

6.4.1.3 Site Visitor

After property transfer, the site will be accessible to the public for limited recreational use in the form of walking tours. Thus, a Site Visitor was evaluated in the risk assessment for IR Site 2. As stated previously, recreational use of the site as a wildlife refuge requires that human activities result in minimal disturbance of wildlife; therefore, it is unlikely that visitors to the site could engage in activities other than walking tours. The Site Visitor for the IR Site 2 risk assessment is evaluated as an individual who

frequents the site once per week throughout the year. The total amount of time spent on site during each visit was limited to two hours to be consistent with the time required for a tour. Furthermore, it was assumed that one hour will be spent outdoors engaged in a walking tour of the site and one hour will be spent indoors participating in a brief presentation given by the Park Ranger/Tour Guide and/or viewing exhibits about the facility.

Both an adult and a child visitor were evaluated in the risk assessment. The Adult Site Visitor was evaluated as an "age-adjusted" individual that frequents the site once per week for 30 years: six years as a 15-kg child and 24 years as a 70-kg adult. The Child Site Visitor was evaluated as a 15-kg individual that frequents the site once per week for six years. Based on U.S. EPA (2004c) risk assessment guidance, carcinogenic health effects were calculated only for the age-adjusted adult visitor and non-carcinogenic health effects were calculated only for the child receptor (i.e., the child is the most sensitive receptor for non-carcinogens; and, the age-adjusted adult is the most sensitive receptor for carcinogens). The following exposure pathways were evaluated in the IR Site 2 risk assessment:

- Ingestion, dermal contact, and inhalation of wind-blown dust containing adsorbed contaminants (evaluated for surface soil);

- Inhalation of VOCs in soil (evaluated for surface and subsurface soil); and,

- External radiation from the presence of radiological constituents in soil (evaluated for surface and subsurface soil).

It is unlikely that the Site Visitors would be exposed to groundwater or surface water at the site; therefore, an assessment of exposure to these media was not conducted for the Site Visitors.

6.4.1.4   Construction/Excavation Worker

A Construction/Excavation Worker was evaluated in the IR Site 2 risk assessment to account for the possibility that construction activities might take place on the site in the future. Based on the future use of the site for recreational purposes, any construction activities would most likely be limited to construction of a small building(s), such as a visitor center or support buildings, and associated utilities (e.g., sewer, water, electricity) and possibly roads. Large-scale construction activities, such as the development of housing, manufacturing or retail facilities, are not anticipated for this site. Thus, the Construction/Excavation Worker was evaluated as a 70-kg adult who works 5 days per week, 8 hours per day, on a combination of aboveground and belowground (i.e., utility installation) construction projects with a total duration of one-half year (26 weeks). Thus, this receptor has an exposure frequency of 130 days per year (5 days/week × 26 weeks/year). Because this receptor has the potential to engage in digging as part of construction or utility installation, and thereby be exposed to contaminants in subsurface soil, the following exposure pathways were evaluated in the IR Site 2 risk assessment:

- Inhalation of dust (from vehicular traffic) containing adsorbed contaminants (evaluated for surface soil);

- Ingestion, dermal contact, and inhalation of VOCs in soil (evaluated for surface and subsurface soil combined);

- External radiation from the presence of radiological constituents in soil (evaluated for surface and subsurface soil combined); and,

- Dermal contact with shallow groundwater.

Dust intake for the Construction/Excavation Worker receptor was evaluated for surface soil rather than combined surface and subsurface soil because vehicular traffic (i.e., construction vehicles) is the mechanism assumed to be responsible for generating dust; and, subsurface soil would not be exposed to vehicular traffic. It is unlikely that this receptor would be exposed to surface water at the site; therefore, an assessment of exposure to this media was not conducted for the Construction/Excavation Worker.

6.4.2 Models for Calculating Intake of Chemical and Radiological Contaminants

This section presents the models that were used to calculate intake of chemical and radiological constituents via the exposure routes relevant for the future site receptors. Models are first presented for calculating intake of chemical contaminants via soil ingestion, soil dermal contact, inhalation of VOCs and dust containing adsorbed contaminants, and dermal contact with water. Then, additional models are presented for calculating intake of radiological contaminants via soil ingestion, inhalation, and external radiation.

6.4.2.1 Incidental Soil Ingestion

Incidental ingestion of soil with chemical contaminants is evaluated for all receptors. The equations for calculating intake via ingestion of chemical contaminants in soil are shown in Equations 6-11 through 6-13. Two forms of the ingestion intake equation are presented because, for the Site Visitor, an age-adjusted intake is calculated for carcinogens (Equations 6-11 and 6-12). As stated previously, carcinogenic health effects were calculated only for the age-adjusted adult visitor and non-carcinogenic health effects were calculated only for the child visitor because the child is the most sensitive receptor for non-carcinogens, and the age-adjusted adult is the most sensitive receptor for carcinogens. The age-adjusted visitor is adopted from the method used by U.S. EPA to calculate residential PRGs for carcinogenic compounds (U.S. EPA, 2004c). The concept of an age-adjusted visitor is plausible if a local resident living near the site frequents the site for a 30-year period. Equation 6-13 is used to calculate intake of non-carcinogens by the Child Site Visitor and it is used to calculate intake of both carcinogens and non-carcinogens by all other receptors. Note that all other receptors are evaluated as an adult and thus the age-adjusted intake adjustment does not apply.

(Equations 6-11 and 6-12) Incidental Ingestion of Soil (Carcinogens/Age-Adjusted Adult Visitor):

$$IFS_{adj} = \frac{ED_c \times IR_c}{BW_c} + \frac{ED_a \times IR_a}{BW_a} \tag{6-11}$$

$$Intake = \frac{IFS_{adj} \times C_{soil} \times EF \times FI \times CF}{AT \times 365 \, days/year} \tag{6-12}$$

(Equation 6-13) Incidental Ingestion of Soil (Non-Carcinogens/Child Visitor; Carcinogens and Non-Carcinogens/All Other Receptors):

$$Intake = \frac{C_{soil} \times IR_{soil} \times EF \times ED \times FI \times CF}{BW \times AT \times 365 \, days/year} \tag{6-13}$$

where: Intake = Amount of chemical at the exchange boundary (mg/kg-day)
     C_{soil} = Chemical concentration in soil (mg/kg)
     IFS_{adj} = Age-adjusted soil ingestion factor (mg-yr/kg-d)
     IR_{soil} = Soil ingestion rate (mg/day)

IR$_a$ = Adult soil ingestion rate (mg/day)
IR$_c$ = Child soil ingestion rate (mg/day)
EF = Exposure frequency (days/year)
ED = Exposure duration (years)
ED$_a$ = Adult exposure duration (years)
ED$_c$ = Child exposure duration (years)
FI = Fraction of soil ingested that is contaminated (unitless)
CF = Conversion factor (10^{-6} kg/mg)
BW = Body weight (kg)
BW$_a$ = Adult body weight (kg)
BW$_c$ = Child body weight (kg)
AT = Averaging time (years)
AT$_c$ = Averaging time for carcinogens (year
AT$_n$ = Averaging time for non-carcinogens (years).

6.4.2.2 Dermal Contact with Soil

Dermal contact with soil containing chemical contaminants is evaluated for all receptors. The equations for calculating intake via dermal contact with soil are shown in Equations 6-14 through 6-17. Again, two forms of the intake equation are shown because an age-adjusted intake is calculated for the age-adjusted adult visitor (Equations 6-14 and 6-15); whereas, intake of non-carcinogens by the child visitor and intake of both carcinogens and non-carcinogens by all other receptors is evaluated using Equations 6-16 and 6-17. In accordance with U.S. EPA (2004c) guidance, dermal contact is not evaluated for VOCs because it is assumed that these constituents will volatilize from soil and thus will not be available for dermal absorption. Instead, VOCs are assessed via the inhalation pathway. Also, only constituents that have a published dermal absorption fraction (ABS$_d$) were evaluated for dermal absorption (U.S. EPA, 2004c).

(Equations 6-14 and 6-15) Dermal Contact with Soil (Carcinogens/Age-Adjusted Adult Visitor):

$$SFS_{adj} = \frac{ED_c \times AF_c \times SA_c}{BW_c} + \frac{ED_a \times AF_a \times SA_a}{BW_a}$$

(6-14)

$$Intake = \frac{SFS_{adj} \times C_{soil} \times ABS_d \times CF \times EV \times EF}{AT \times 365 \, days \, / \, year}$$

(6-15)

(Equations 6-16 and 6-17) Dermal Contact with Soil (Non-Carcinogens/Child Visitor; Carcinogens and Non-Carcinongens/All Other Receptors):

$$DA_{event} = C_{soil} \times AF \times ABS_d \times CF$$

(6-16)

$$Intake = \frac{DA_{event} \times EV \times ED \times EF \times SA}{BW \times AT \times 365 \, days \, / \, year}$$

(6-17)

where: Intake = Dermally absorbed dose (mg/kg-day)
DA$_{event}$ = Absorbed dose per event per area of skin exposed (mg/cm^2-event)
SFS$_{adj}$ = Age adjusted factor (mg-yr/kg-event)
C$_{soil}$ = Contaminant concentration in soil (mg/kg)
EV = Event frequency (events/day)
ED = Exposure duration (years)
ED$_a$ = Adult exposure duration (years)

ED$_c$ = Child exposure duration (years)
EF   = Exposure frequency (days/year)
SA   = Skin surface area available for contact (cm^2)
SA$_a$ = Skin surface area available for contact, adult (cm^2)
SA$_c$ = Skin surface area available for contact, child (cm^2)
BW   = Body weight (kg)
BW$_a$ = Body weight, adult (kg)
BW$_c$ = Body weight, child (kg)
AT   = Averaging time (years)
AT$_n$ = Averaging time for non-carcinogens (years)
AT$_c$ = Averaging time for carcinogens (years)
AF   = Soil to skin adherence factor, chemical-specific (mg/cm^2-event)
AF$_a$ = Soil to skin adherence factor, adult (mg/cm^2-event)
AF$_c$ = Soil to skin adherence factor, child (mg/cm^2-event)
ABS$_d$ = Dermal absorption fraction, chemical-specific (unitless)
CF   = Conversion factor (10^{-6} kg/mg).

6.4.2.3   Inhalation of VOCs in Soil

Receptors can be exposed to VOCs in soil through inhalation because these chemicals can volatilize from the soil into outdoor air. Thus, inhalation of VOCs in soil is evaluated for all receptors. Unlike ingestion and dermal exposures, intake of VOCs via inhalation is not calculated, because the conversion from concentration in air to internal dose (intake) is not always appropriate due to portal-of-entry effects and differences in absorption in the gut versus the lungs (U.S. EPA, 1996). Rather, an exposure concentration adjusted for exposure frequency, duration, and time (C$_{air}$) is calculated using Equation 6-18; C$_{air}$ is the concentration of a chemical that a receptor is exposed to in outdoor air during the exposure duration. C$_{air}$ is used with chemical specific toxicity data to calculate cancer and/or non-cancer health effects, as explained in Section 6.4.4.3.

C$_{air}$ is a predicted concentration that is calculated based on the measured concentration of the chemical in soil and a chemical-specific volatilization factor (VF) which expresses the ratio of the concentration of a VOC in air to the concentration of the chemical in soil at equilibrium (U.S. EPA, 2001a). The equations for calculating VF for all receptors except the Construction/Excavation Worker are shown in Equations 6-19 through 6-21. As shown in these equations, calculation of VF requires information describing the chemical (e.g., D$_i$, D$_w$, H', K$_{oc}$) and information describing the site (e.g., A$_{site}$, Q/C, ρ_b), thus, a separate VF is calculated for each VOC. Equations 6-22 through 6-24 are used to calculate the VF for evaluating inhalation exposure for the Construction/Excavation Worker. These equations differ only slightly from those used to calculate VF for the other receptors. For example, the VF equation is identical except that it includes a dispersion correction factor, F$_D$, to account for sites that are smaller than the default area of 0.5 acre. In addition, the constants A, B, and C used to calculate a site-specific Q/C term are the same for all sites rather than varying by location.

(Equation 6-18) Inhalation of VOCs in Soil:

$$C_{air} = \frac{C_{soil} \times \dfrac{1}{VF} \times EF \times ED \times ET}{AT \times 365 \, days / years \times 24 \, hrs / d} \tag{6-18}$$

where: C$_{soil}$ = Contaminant concentration in soil (mg/kg)
        C$_{air}$ = Adjusted exposure concentration in outdoor air (mg/m^3)

EF   = Exposure frequency (days/year)
ED   = Exposure duration (years)
ET   = Exposure time per day (hours/day)
AT_n = Averaging time for non-carcinogens (years)
AT_c = Averaging time for carcinogens (years)
VF   = Soil to air volatilization factor, chemical-specific (m^3/kg).

(Equations 6-19 through 6-21) Volatilization Factor (all Receptors Except Construction/Excavation Worker):

$$VF = \frac{(Q/C) \times (3.14 \times D_A \times T)^{1/2} \times 10^{-4}(m^2/cm^2)}{(2 \times \rho_b \times D_A)} \quad (6\text{-}19)$$

where:

$$Q/C = A \times exp\left[\frac{(\ln A_{site} - B)^2}{C}\right] \quad (6\text{-}20)$$

$$D_A = \frac{\left[\left(\left(\theta_a^{10/3} \times D_i \times H'\right) + \left(\theta_w^{10/3} \times D_w\right)\right)/n^2\right]}{\left(\left(\rho_b \times k_d\right) + \theta_w + \left(\theta_a \times H'\right)\right)} \quad (6\text{-}21)$$

and where: Q/C = Inverse of the mean concentration at the center of a square source (g/m^2 per kg/m^3)
          DA   = Apparent diffusivity, chemical-specific (cm^2/s)
          T   = Exposure interval, equal to exposure duration(s)
          A_{site} = Area of the site (acres)
A, B, and C = Constants depending on the location of the site
          ρ_b = Dry soil bulk density (g/cm^3)
          θ_a = Air-filled soil porosity (L_{air}/L_{soil}) ($\theta_a = n - \theta_w$)
          θ_w = Water-filled soil porosity (L_{water}/L_{soil})
          n   = Total soil porosity (L_{pore}/L_{soil})
          D_i = Diffusivity in air, chemical-specific (cm^2/s)
          D_w = Diffusivity in water, chemical-specific (cm^2/s)
          H'   = Dimensionless Henry's Law Constant, chemical-specific
          K_d = Soil-water partition coefficient (cm^3/g), (for organic chemicals, $K_d=K_{oc} \times f_{oc}$)
          K_{oc} = Soil organic carbon partition coefficient, chemical-specific (cm^3/g)
          f_{oc} = Fraction organic carbon in soil (g/g).

(Equations 6-22 through 6-24) Volatilization Factor (Construction/Excavation Worker):

$$VF = \left[\frac{(3.14 \times D_A \times T)^{0.5}}{2 \times \rho_b \times D_A}\right] \times 10^{-4} m^2/cm^2 \times Q/C \times \frac{1}{F_D} \quad (6\text{-}22)$$

where:

$$Q/C = A \times exp\left[\frac{(\ln A_{site} - B)^2}{C}\right] \quad (6\text{-}23)$$

$$D_A = \frac{\left[\left(\left(\theta_a^{10/3} \times D_i \times H'\right) + \left(\theta_w^{10/3} \times D_w\right)\right)/n^2\right]}{\left(\left(\rho_b \times k_d\right) + \theta_w + \left(\theta_a \times H'\right)\right)}$$ (6-24)

and where: VF = Volatilization factor (m³/kg)
Q/C_{sa} = Inverse of the mean concentration at the center of a 0.5-acre square source (g/m² per kg/m³)
DA = Apparent diffusivity, chemical-specific (cm²/s)
T = Total time over which construction occurs, site-specific (s)
A_c = Area of the site (acres)
A, B, and C = Constants (A = 2.4538; B = 17.5660; and C = 189.0426)
ρ_b = Dry soil bulk density (g/cm³)
F_D = Dispersion correction factor (unitless)
θ_a = Air-filled soil porosity (L_{air}/L_{soil}) ($\theta_a = n - \theta_w$)
θ_w = Water-filled soil porosity (L_{water}/L_{soil})
n = Total spoil porosity (L_{pore}/L_{soil})
D_i = Diffusivity in air, chemical-specific (cm²/s)
D_w = Diffusivity in water, chemical-specific (cm²/s)
H' = Dimensionless Henry's Law Constant, chemical-specific
K_d = Soil-water partition coefficient (cm³/g), (for organic chemicals, $K_d = K_{oc} \times f_{oc}$)
K_{oc} = Soil organic carbon partition coefficient, chemical-specific (cm³/g)
f_{oc} = Fraction organic carbon in soil (g/g).

6.4.2.4 Inhalation of Non-Volatile Compounds in Surface Soil (Fugitive Dust)

In addition to inhaling VOCs that volatilize from soil to outdoor air, receptors can also inhale non-volatile compounds present in soil if the soil is subject to wind erosion/suspension or other mechanisms such as vehicle traffic that can produce fugitive dust. Thus, inhalation of non-volatile compounds in surface soil is evaluated for all receptors. Only non-volatile compounds in surface soil are evaluated for this exposure pathway because subsurface soils are not susceptible to wind erosion/suspension or vehicular traffic. As is the case for inhalation of VOCs, inhalation intake of non-volatile compounds is not calculated. Rather, an adjusted exposure concentration (C_{air}) is calculated using Equation 6-25. As mentioned above, C_{air} is used with chemical specific toxicity data to calculate cancer and/or non-cancer health effects (see Section 6.4.4.3). C_{air} for the fugitive dust exposure pathway is a predicted concentration in air that is calculated based on the measured concentration of the chemical in soil and a site-specific particulate emission factor (PEF) (U.S. EPA, 2001a). The PEF expresses the ratio of the concentration of a non-volatile compound in air to the concentration of the chemical in soil. The equations for calculating PEF for all receptors except the Construction/Excavation Worker are shown in Equations 6-24 and 6-27. The PEF for these receptors is based on wind erosion/suspension and other dust-generating mechanisms such as vehicle traffic are assumed to be negligible. As shown in these equations, calculation of PEF requires information describing the site (e.g., A_{site}, Q/C, U_t, U_m, V); thus, a single PEF is calculated for all non-volatile compounds. Equations 6-28 and 6-29 are used to calculate the PEF for evaluating inhalation exposure for the Construction/Excavation Worker. These equations differ significantly from the equations used to calculate the PEF for the other receptors because vehicle traffic is considered the primary dust-generating mechanism. Wind erosion/suspension is considered to be negligible for the Construction/Excavation Worker in comparison to vehicle traffic. Also, the same values of the constants A, B, and C are used to calculate a site-specific Q/C term regardless of the location of the site (A, B, and C vary by location when calculating Q/C for the windblown PEF).

(Equation 6-25) Inhalation of Non-Volatile Compounds in Surface Soil:

$$C_{air} = \frac{C_{soil} \times \dfrac{1}{PEF} \times EF \times ED \times ET}{AT \times 365\,days\,/\,year \times 24\,hr\,/\,d}$$

(6-25)

where: C_{air} = Adjusted exposure concentration in outdoor air (mg/m^3)
EF = Exposure frequency (days/year)
ED = Exposure duration (years)
ET = Exposure time per day (hours/day)
AT_n = Averaging time for non-carcinogens (years)
AT_c = Averaging time for carcinogens (years)
PEF = Particulate emission factor (m^3/kg).

(Equations 6-26 and 6-27) Particulate Emission Factor (all Receptors Except Construction/Excavation Worker):

$$PEF = Q\,/\,C \times \frac{3600\,s\,/\,hr}{0.036 \times (1-V) \times \left(\dfrac{U_m}{U_t}\right)^3 \times F(x)}$$

(6-26)

where:

$$Q\,/\,C = A \times exp\left[\frac{(ln\,A_{site} - B)^2}{C}\right]$$

(6-27)

and where: V   = Fraction of vegetative cover (unitless)
U_m = Mean annual wind speed (m/s)
U_t = Equivalent threshold value of the windspeed at 7m (m/s)
F(x)   = Function dependent on U_m/U_t derived (unitless)
A_{site} = Area of the site (acres)
A, B, and C = Constants depending on the location of the site.
Q/C   = Inverse of the mean concentration at the center of a square source (g/m^2 per kg/m^3).

(Equations 6-28 and 6-29) Particulate Emission Factor (Construction/Excavation Worker):

$$PEF = Q\,/\,C_{sr} \times \frac{1}{F_D} \times \left[\frac{T \times A_R}{556 \times \left(\dfrac{W}{3}\right)^{0.4} \times \dfrac{(365\,d\,/\,yr - p)}{365\,d\,/\,yr} \times \sum VKT}\right]$$

(6-28)

where:

$$Q\,/\,C_{sr} = A \times exp\left[\frac{(ln\,A_s - B)^2}{C}\right]$$

(6-29)

and where: PEF = Vehicular traffic particulate emission factor (m^3/kg)

Q/C_{sr} = Inverse of 1-h average air concentration along a straight road segment bisecting a square site (g/m^2-s per kg/m^3) where,

As = Site area, site-specific (acres)

A, B, and C = Constants (A = 12.9351; B = 5.7383; and C = 71.7711)

T = Total time over which construction occurs, site-specific (s)

A_R = Surface area of contaminated road segment, equal to length × width of road (m^2)

W = Mean vehicle weight (tons)

P = Number of days with at least 0.01 inches of precipitation, site-specific (days/year)

$\sum$VKT = sum of fleet vehicle kilometers traveled during the exposure duration, site-specific (km).

6.4.2.5   Dermal Contact with Surface Water and Groundwater

Dermal contact with contaminants in surface water and groundwater is considered a plausible exposure route for the Ranger/Habitat Restoration Supervisor because habitat restoration may involve work in or adjacent to the wetland ponds at the site or digging holes that extend to a depth sufficient to reach shallow groundwater. Similarly, dermal contact with groundwater is considered to be a plausible pathway for the Construction/Excavation Worker because it may be necessary for this receptor to work in a trench that intersects the shallow water table if installation and/or maintenance of underground utilities at the site are required. There are currently no known underground utilities at IR Site 2. If a building(s) (e.g, visitor center) is constructed on site in the future, there would likely be underground utilities associated with this building (i.e., sewer, water). At this time, it is not known whether or not buildings with utilities will be constructed at the site in the future. Dermal exposure to contaminants in surface water and groundwater is not evaluated for any of the other receptors.

Equation 6-30 is used to calculate dermal intake of chemicals in water (U.S. EPA, 2004c). Calculation of dermal intake requires calculation of an absorbed dose per event (DA $_{event}$) which differs for organic chemicals (Equations 6-31 and 6-32) and inorganic chemicals (Equation 6-34). DA $_{event}$ is a chemical-specific parameter, and, it also depends on the event duration; thus, DA $_{event}$ is calculated for each chemical. Equations for calculating chemical-specific parameters that are included in these equations, including K_p, FA, t_{event}, B, Tau (τ)$_{event}$, and t*, are described in U.S. EPA (2004c).

(Equation 6-30) Dermal Intake of Contaminants in Water:

$$\text{Intake} = \frac{DA_{event} \times EV \times ED \times EF \times SA}{BW \times AT \times 365 \text{days} / \text{year}}$$

(6-30)

(Equations 6-31 and 6-32) Absorbed Dose for Organic Chemicals:

$$\text{If } t_{event} \leq t^*, \quad DA_{event} = 2 \times FA \times K_p \times C_w \times \sqrt{\frac{6 \times \tau_{event} \times t_{event}}{\pi}}$$

(6-31)

$$\text{If } t_{event} > t^*, \quad DA_{event} = FA \times K_p \times C_w \left[\frac{t_{event}}{1+B} + 2(\tau_{event}) \left[\frac{1 + 3B + 3B^2}{(1+B)^2} \right] \right]$$

(6-32)

where: Intake = Dermally absorbed dose (mg/kg-day)

DA $_{event}$ = Absorbed dose per event per area of skin exposed (mg/cm^2-event)
C$_w$ = Contaminant concentration in water (mg/cm^3)
EV = Event frequency (events/day)
ED = Exposure duration (years)
EF = Exposure frequency (days/year)
SA = Skin surface area available for contact (cm^2)
BW = Body weight (kg)
AT = Averaging time (years)
FA = Fraction absorbed water, chemical specific (dimensionless)
t$_{event}$ = Event duration (hours/event)
K$_p$ = Skin permeability coefficient of compound in water, chemical-specific (cm/hour)
B = Dimensionless ratio of the permeability of the stratum corneum relative to the permeability across the viable epidermis, chemical-specific (dimensionless)
Tau (τ)$_{event}$ = lag time per event, chemical-specific (hours/event)
t* = time to reach steady state (hr), chemical-specific. $2.4 \times \tau_{event}$

(Equation 6-33) Absorbed Dose for Inorganic Chemicals:

$$DA_{event} = K_p \times C_w \times t_{event} \tag{6-33}$$

6.4.2.6   Exposure to Radionuclides

Receptors at IR Site 2 may be exposed to radionuclides that are present in soil via soil ingestion, inhalation of fugitive dust, and external radiation. In general, radionuclides are not volatile compounds (radon is an exception); therefore, volatilization from soil is not a concern unless a volatile radionuclide is present. Similarly, dermal absorption of radionuclides is not evaluated because these constituents are considered to have negligible absorption across the skin. External radiation is an exposure route that is unique to radionuclides and results from emitted radiation (i.e., alpha, beta, gamma particles/rays) contacting the external surface of the body. Furthermore, all radionuclides are considered to be Class A carcinogens and as such are evaluated for their cancer-causing potential; non-cancer effects are not evaluated.

6.4.2.6.1   Incidental Ingestion of Soil Containing Radionuclides

Equations 6-34 and 6-35 are used to calculate intake of radionuclides via soil ingestion by the age-adjusted adult visitor (U.S. EPA, 2000a). Equation (6-36) is used to calculate intake of radionuclides by all other receptors. Unlike intake of chemicals, intake of radionuclides is expressed as the total intake of the constituent over a lifetime, in pCi; consequently, body weight and averaging time are not used to calculate intake. In addition, because radionuclides undergo decay, a decay constant, λ, is incorporated into the equation to account for diminishing concentrations throughout the exposure duration.

(Equations 6-34 and 6-35) Incidental Ingestion of Soil (Age-Adjusted Adult Visitor):

$$IFS_{adj} = \frac{(ED_c \times IR_c) + (ED_a \times IR_a)}{ED} \tag{6-34}$$

$$Intake = \frac{IFS_{adj} \times C_{soil} \times (1 - e^{-\lambda t}) \times EF \times ED \times FI \times CF}{t \times \lambda} \tag{6-35}$$

(Equation 6-36) Incidental Ingestion of Soil (All Other Receptors):

$$\text{Intake} = \frac{C_{soil} \times (1 - e^{-\lambda t}) \times IR_{soil} \times EF \times ED \times FI \times CF}{t \times \lambda} \qquad (6\text{-}36)$$

where: Intake = Lifetime intake of radionuclide from incidental ingestion (pCi)
   C_{soil} = Chemical concentration in soil (pCi/kg)
   λ = Decay constant, radionuclide specific $(\text{yr})^{-1}$
   t = Time, equal to exposure duration (yr)
   IFS_{adj} = Age-adjusted soil ingestion factor (mg/d)
   IR_{soil} = Soil ingestion rate (mg/day)
   IR_a = Adult soil ingestion rate (mg/day)
   IR_c = Child soil ingestion rate (mg/day)
   EF = Exposure frequency (days/year)
   ED = Exposure duration (years)
   ED_a = Adult exposure duration (years)
   ED_c = Child exposure duration (years)
   FI = Fraction of soil ingested that is contaminated (unitless)
   CF = Conversion factor (10^{-6} kg/mg).

6.4.2.6.2 Inhalation of Radionuclides Adsorbed to Fugitive Dust

Equations 6-37 and 6-38 are used to calculate inhalation of radionuclides associated with fugitive dust by the age-adjusted adult visitor (U.S. EPA, 2000a; ORNL, 2005). Equation 6-39 is used to calculate inhalation of radionuclides by all other receptors. As is the case for inhalation of chemicals associated with fugitive dust, the PEF is used to calculate a concentration of the radionuclide in air based on the measured concentration in soil. The PEF is calculated using Equations 6-26 and 6-27 for all receptors except Construction/Excavation Worker, and Equations 6-28 and 6-29 for the Construction/Excavation Worker, as presented previously. Unlike chemical contaminants, it is necessary to calculate an inhalation intake for radionuclides (in pCi) because the toxicity critera (i.e., cancer slope factor) for radionuclides is expressed in units of pCi^{-1} (as discussed previously, inhalation intake of chemical contaminants is not calculated). In addition, the exposure time is apportioned into the time spent indoors and outdoors, and an indoor dilution factor, DF_i, is applied to the time spent indoors to account for reduced concentrations in indoor air.

(Equation 6-37 and 6-38) Inhalation of Fugitive Dust (Age-Adjusted Adult Visitor):

$$InR_{adj} = \frac{(ED_c \times InR_c) + (ED_a \times InR_a)}{ED} \qquad (6\text{-}37)$$

$$\text{Intake} = \frac{C_{soil} \times (1 - e^{-\lambda t}) \times (\frac{1}{PEF}) \times InR_{adj} \times EF \times ED \times (ET_o + (ET_i \times DF_i))}{t \times \lambda} \qquad (6\text{-}38)$$

(Equation 6-39) Inhalation of Fugitive Dust (All Other Receptors):

$$\text{Intake} = \frac{C_{soil} \times (1 - e^{-\lambda t}) \times (\frac{1}{PEF}) \times InR \times EF \times ED \times (ET_o + (ET_i \times DF_i))}{t \times \lambda} \qquad (6\text{-}39)$$

where: Intake = Lifetime intake of radionuclide from inhalation (pCi)
      C_{soil} = Concentration in soil (pCi/Kg)
      λ = Decay constant, radionuclide specific $(yr)^{-1}$
      t = Time, equal to exposure duration (yr)
      PEF = Particulate emission factor (m^3/kg)
      InR = Inhalation rate (m^3/day)
      InR_a = Adult inhalation rate (m^3/day)
      InR_c = Child inhalation rate (m^3/day)
      InR_{adj} = Age-adjusted inhalation factor (m^3/d)
      EF = Exposure frequency (days/year)
      ED = Exposure duration (years)
      ED_a = Adult exposure duration (years)
      ED_c = Child exposure duration (years)
      ET_o = Exposure time fraction, outdoors (unitless)
      ET_i = Exposure time fraction, indoors (unitless)
      DF_i = Dilution factor for indoor air inhalation=0.4 (unitless).

6.4.2.6.3 External Radiation from Exposure to Radionuclides in Soil

Equation 6-40 is used to calculate external radiation from radionuclides in soil (U.S. EPA, 2000a; ORNL, 2005). The same equation is used for all receptors. As with inhalation of radionuclides, an apportioning factor and a shielding factor is applied to account for time spent indoors, which results in a reduced level of external radiation.

$$\text{Intake} = \frac{C_{soil} \times (1 - e^{-\lambda t}) \times EF \times ED \times \dfrac{yr}{365\,d} \times ACF \times (ET_o + (ET_i \times GSF)) \times CF}{t \times \lambda} \qquad (6\text{-}40)$$

where: Intake = Lifetime intake of radionuclide from inhalation (pCi-yr/g)
      C_{soil} = Concentration in soil (pCi/kg)
      λ = Decay constant, radionuclide specific $(yr)^{-1}$
      t = Time, equal to exposure duration (yr)
      EF = Exposure frequency (days/year)
      ED = Exposure duration (years)
      ACF = Area correction factor (unitless)
      ET_o = Exposure time fraction, outdoors (unitless)
      ET_i = Exposure time fraction, indoors (unitless)
      GSF = Gamma shielding factor = 0.4 (unitless)
      CF = Conversion factor $(10^{-3}\ kg/g)$.

6.4.3 Exposure, Chemical-Specific, and Site Parameters Used to Calcualate Intake

Models that are used to calculate exposure include numerous parameters that must be carefully defined before accurate intake estimates can be determined. These include exposure parameters that define the receptors and their activity patterns, chemical-specific parmameters, and site parameters. This section briefly describes each type of parameter and defines the values that were selected for the risk assessment.

6.4.3.1   Exposure Point Concentrations

Estimates of contaminant concentrations at points of potential human exposure are necessary for evaluating contaminant intakes by potentially exposed individuals. Such long-term exposure point concentrations (EPCs) are developed from short-term monitoring data, with the underlying assumption that the monitoring data are representative of, or simulate, future conditions at the site. EPCs are assumed to remain constant for an indefinite period of time, and it is generally assumed that no abiotic or biotic degradation mechanisms will lower these concentrations.

EPCs were developed to model exposures under the reasonable maximum exposure (RME) scenario, which is defined as the highest exposure that is reasonably expected to occur at a site (U.S. EPA, 1989a). RME estimates are calculated using a combination of upper bound values for exposure parameters (e.g., ingestion rate and inhalation rate) and an estimate of the mean exposure point concentration. U.S. EPA defines the EPC used to estimate the RME as the 95th UCL on the arithmetic mean, or the maximum detected concentration, whichever is lower. To estimate the 95% UCL, appropriate statistical tests were applied to examine the distribution of the data to determine the most appropriate method(s) for calculating the 95% UCL. Criteria outlined in U.S. EPA (2002a) document *Calculating Exposure Point Concentrations at Hazardous Waste Sites* were used to determine the most appropriate statistical method for calculating an exposure point concentration that is representative of the average concentration (i.e., UCL). Appendix G provides a comprehensive discussion of methods used to compute UCLs; and, Tables 6-3 to 6-6 list the summary statistics, including the UCLs, that were calculated for each constitutent in each media.

6.4.3.2   Exposure Parameters

Exposure parameters include the variables used in the intake models that describe the exposed individuals, including the contact rate (e.g., ingestion rate, skin surface area), exposure frequency and duration, and body weight. As stated previously, RME exposure estimates are calculated using upper bound values for exposure parameters. Values for the exposure parameters were obtained from a number of sources, as listed below:

- Various standard references for human health exposure factors, including:

  - U.S. EPA, 1989a. Risk Assessment for Superfund, Volume I Human Health Evaluation Manual (Part A);

  - U.S. EPA, 1991a. Human Health Evaluation Manual, Supplemental Guidance: Standard Default Exposure Factors;

  - U.S. EPA, 1997a-c. Exposure Factors Handbook, Volumes I-III;

  - U.S. EPA, 2004c. Human Health Evaluation Manual, Part E, Supplemental Guidance for Dermal Risk Assessment;

  - U.S. EPA, 2001a. Supplemental Guidance for Developing Soil Screeing Levels for Superfund Sites, Interim Guidance;

  - U.S. EPA, 2000a. Soil Screening Guidance for Radionuclides: User's Guide.

  - U.S. EPA, 2004b. User's Guide and Technical Background Document for U.S. EPA's Region 9 Preliminary Remediation Goals Table;

  - DTSC, 1999a. Preliminary Endangerment Assessment Guidance Manual.

- Per request of DTSC, exposure parameters for the ranger and restoration supervisor were taken from the BRA for NWS Seal Beach, Sites 4, 5, and 6 (Bechtel, 2001).

- A combination of sources was consulted to derive appropriate exposure parameters for the Construction/Excavation Worker because U.S. EPA and DTSC have not developed standard exposure assumptions for this scenario, including:

  - Massachusetts Department of Environmental Protection (2004): S-3 Standards, Construction/Excavation Potential.

  - Michigan Department of Environmental Quality (2003): Storage Tank Division, Part 213 - Tier 1 Criteria for Utility Worker, Groundwater Direct Contact; and

  - ORNL (2005): Risk Assessment Information System.

Exposure parameters are described on Tables 6-16 through 6-20 for the receptors evaluated in the BRA, including: the Park Ranger/Tour Guide (Table 6-16), the Ranger/Restoration Supervisor (Table 6-17), the Adult Site Visitor (Table 6-18) and the Child Site Visitor (Table 6-19), and the Construction/Excavation Worker (Table 6-20).

6.4.3.3 Site and Chemical-Specific Parameters

Models used to describe intake/exposure include a number of chemical-specific parameters in addition to the EPC. Chemical-specific parameters include variables required to calculate the dermally absorbed dose from exposure to contaminants in soil and groundwater (e.g., ABS_d, ABS_{GI}, K_p), and parameters recquired to calculate the volatilization factor (VF) for evaluating VOC inhalation exposures (e.g., D_i, D_w, H'). For radionuclides, it is also necessary to define the decay constant (λ) for the radionuclides included in the risk assessment. Table 6-21 provides the values of the chemical-specific parameters for each constituent that was evaluated in the BRA.

In addition to chemical-specific parameters, a number of site parameters (e.g., wind speed, soil moisture content) are required to calculate VF and PEF, which are used to evaluate VOC and dust inhalation exposures. Both the VF and the PEF depend on the size of the site; therefore, VF and PEF were calculated for each of the two general site areas (i.e., landfill and wetland). The VF and PEF for the landfill is based on an area of 77 acres; whereas, the VF and PEF for the wetland area are basd on an area of 33 acres. Additionally, the VF and PEF for the Construction/Excavation Worker are calculated differently than the VF and the PEF for the other receptors because the Construction/Excavation Worker is assumed to be exposed to dust and VOCs as a result of vehicle traffic rather than wind suspension; therefore, additional site parameters are required to define these parameters. Table 6-22 provides the values of the site parameters used to compute VF and PEF for all receptors except the Construction/Excavation Worker; Table 6-23 provides the values of the site parameters used to compute VF and PEF for the Construction/Excavation Worker.

6.4.4 Models for Calculating Risk and Hazard

Carcinogens are agents that induce cancer. Potential carcinogenic effects are expressed as the probability that an individual will develop cancer from a lifetime of exposure to a contaminant, according to the following general relationship (U.S. EPA, 1989a):

$$\text{Cancer Risk} = \text{Intake} \times \text{Cancer Slope Factor} \qquad (6-41)$$

The cancer slope factor (CSF) is the numerical estimate of cancer potency of a carcinogenic contaminant. The CSF defines the cancer risk due to constant lifetime exposure (24 hours a day for 365 days a year) to one unit of carcinogen, generally in units of risk per mg/kg/day. CSFs are derived by calculating the 95% UCL on the slope of the linearized portion of the dose-response curve obtained from a multistage

(nonlinear) cancer model of a carcinogen. This is a conservative approach and is likely to overestimate the level of risk, as the actual risk is expected to be between zero and the calculated value. Carcinogenic slope factors assume no threshold for effects (e.g., cumulative effects of exposure to multiple carcinogens) and that exposure to any concentration of a carcinogen has the potential to produce a carcinogenic effect. Cancer risks from exposure to multiple carcinogens and multiple pathways are assumed to be additive. To obtain an estimate of total risk from all carcinogens at the site, cancer risks are summed across all exposure pathways for potential carcinogens of concern.

Noncarcinogenic effects were evaluated using reference doses (RfDs) developed by U.S. EPA. RfDs are expressed as acceptable daily doses in milligrams of contaminant per kilogram of body weight per day (mg/kg-day). The RfD is a health-based criterion based on the assumption that thresholds exist for noncarcinogenic toxic effects (e.g., liver or kidney damage) over a length of time of exposure (i.e., chronic versus acute). In general, the chronic RfD is an estimate (with uncertainty spanning perhaps an order of magnitude) of a daily exposure to the human population (including sensitive subgroups) that is likely to be without an appreciable risk of deleterious effects during a lifetime of exposure (U.S. EPA, 1989a). Chronic RfDs are specifically developed to be protective against long-term exposure to a contaminant between seven years and a lifetime. Potential health effects related to chronic exposure to noncarcinogenic compounds were assessed by calculating a HQ for each chemical. An HQ is derived by dividing the estimated daily intake by a contaminant-specific chronic RfD, as in the following general relationship (U.S. EPA, 1989a):

$$HQ = \frac{\text{Intake}}{\text{RfD}} \qquad (6\text{-}42)$$

Generally, contaminant-specific HQs are summed to derive pathway-specific hazard index (HI) values. This approach may result in a situation where HI values exceed unity (i.e., one), even when no contaminant-specific HQs exceed one, indicating adverse systemic health effects would be expected to occur only if the receptor were exposed to multiple contaminants simultaneously. In this case, contaminants are sometimes segregated by effect and mode of action on a target organ, resultiung in separate HI values for each effect (U.S. EPA, 1989a). If any of the separate HI values exceed one, it is assumed that adverse, noncarcinogenic health effects are possible.

Specific forms of the above equations are presented below for each exposure route that was evaluated in the risk assessment, including: ingestion, dermal contact, inhalation, and external radiation.

6.4.4.1 Ingestion of Soil

Cancer risks and non-cancer HQs for ingestion of soil containing chemical contaminants is evaluated according to the relationships shown in Equations 6-43 and 6-44.

$$\text{Risk} = \text{Intake} \times \text{CSF}_0 \qquad (6\text{-}43)$$

$$HQ = \frac{\text{Intake}}{\text{RfD}_0} \qquad (6\text{-}44)$$

where: Intake = Amount of chemical at the exchange boundary/gut (mg/kg-day)
$\quad\quad$ CSF$_0$ = Oral cancer slope factor, chemical-specific (mg/kg-day)$^{-1}$
$\quad\quad$ RfD$_0$ = Oral reference dose, chemical-specific (mg/kg-day).

For radionuclides, cancer risk is calculated according to Equation 6-45:

$$Risk = Intake \times CSF_o \qquad (6\text{-}45)$$

where: Intake = Lifetime intake of radionuclide from ingestion (pCi)
CSF_o = Radionuclide cancer slope factor, radionuclide-specific $(pCi)^{-1}$.

6.4.4.2 Dermal Contact with Soil and Water

Cancer risk and non-cancer HQs for dermal contact with soil and water containing chemical contaminants is evaluated according to the relationships shown in Equations 6-46 through 6-47.

$$Risk = Intake \times CSF_d \qquad (6\text{-}46)$$

$$HQ = \frac{Intake}{RfD_d} \qquad (6\text{-}47)$$

As discussed in U.S. EPA (2004c), intake from dermal contact describes the absorbed dose; however, toxicity parameters such as RfD and CSF are derived from administered doses. Therefore, to characterize risk from the dermal exposure pathway, adjustment of the oral toxicity factor is required to represent an absorbed dose rather than an administered dose. This adjustment accounts for the absorption efficiency in the "critical study", which forms the basis for the RfD or CSF. In the case where oral absorption in the critical study is essentially complete, the absorbed dose is equal to the administered dose and no toxicity adjustment is required. When oral absorption of a chemical in the critical study is poor, the absorbed dose is much less than the administered dose. In this case, toxicity factors based on the absorbed dose should be adjusted to account for the difference in the absorbed dose relative to the administered dose. U.S. EPA (2004c) recommends adjusting the oral toxicity factor when: (1) the toxicity value derived from the critical study is based on an administered dose; and (2) scientifically defensible data are available that demonstrate that the gastrointestinal (GI) absorption of the chemical in question, from a medium similar to the one employed in the critical study, is significantly less than 100%. U.S. EPA (2004c) provides GI absorption values for several chemicals, including both organic and inorganic chemicals. For chemicals not listed by U.S. EPA (2004c), an ABS of 100% was assumed. Toxicity parameters that reflect absorbed dose rather than administered dose are calculated using Equations 6-48 and 6-49 for carcinogens and non-carcinogens, respectively (U.S. EPA, 2004c).

$$CSF_d = \frac{CSF_o}{ABS_{GI}} \qquad (6\text{-}48)$$

$$RfD_d = RfD_o \times ABS_{GI} \qquad (6\text{-}49)$$

where: CSF_d = Cancer slope factor for dermal contact, chemical-specific $(mg/kg\text{-}day)^{-1}$
ABS_{GI} = Gastrointestinal absorption factor, chemical-specific (dimensionless)
RfD_d = Reference dose for dermal contact, chemical-specific (mg/kg-day).

6.4.4.3 Inhalation (VOCs and Fugitive Dust)

Cancer risks and non-cancer HQs from inhalation of contaminants in air and dust is evaluated according to the relationships shown in Equations 6-50 and 6-51.

$$Risk = C_{air} \times URF \times CF \qquad (6\text{-}50)$$

$$HQ = \frac{C_{air}}{RfC} \tag{6-51}$$

where: URF = Inhalation unit risk factor, chemical-specific $(\mu g/m^3)^{-1}$
      CF   = Conversion factor $(10^{-3} \mu g/mg)$
      RfC = Inhalation reference concentration, chemical-specific (mg/m^3).

As of 1991, U.S. EPA toxicity databases such as IRIS no longer present RfDs or CSFs for the inhalation route (U.S. EPA, 2004b). These values have been replaced with the reference concentration (RfC) for non-carcinogenic effects and unit risk factors (URF) for carcinogenic effects. Therefore, cancer risk and non-cancer HQs are calculated by comparing estimated air concentrations of volatiles/particulates, adjusted for exposure frequency, duration, and time (as described previously), with these inhalation toxicity values.

For radionuclides, cancer risk is calculated according to Equation 6-52:

$$Risk = Intake \times CSF \tag{6-52}$$

where: Intake = Lifetime intake of radionuclide from inhalation (pCi)
      CSF   = Inhalation slope factor for radionuclides, radionuclide-specific $(pCi)^{-1}$.

6.4.4.4 External Radiation

For radionuclides, cancer risk from external radiation is calculated according to Equation 6-53:

$$Risk = Intake \times CSF \tag{6-53}$$

where: Intake = Lifetime intake of radionuclide from external radiation (pCi)
      CSF   = External radiation slope factor for radionuclides, radionuclide-specific
              (1/yr per pCi/g).

6.4.5 Toxicity Assessment

In the toxicity assessment, data were compiled for each constituent that characterize the relationship between the magnitude of exposure to the constituent and the nature and magnitude of adverse health effects that may result from each exposure. Two broad categories of health effects were calculated in this risk assessment: carcinogenic and noncarcinogenic. Therefore, toxicity data were compiled that describe a constituent's tendency to illicit cancer and non-cancer health effects. Toxicity values and exposure criteria are generally developed based on the threshold approach for noncarcinogenic effects and the non-threshold approach for carcinogenic effects.

For this assessment, toxicity criteria were selected according to the U.S. EPA (2003a) guidance that recommends a hierarchy of human health toxicity values for use in risk assessments at Superfund sites. The hierarchy is as follows: (1) U.S. EPA's Integrated Risk Information System (IRIS); (2) U.S. EPA's (Office of Research and Development, National Center for Environmental Assessment, Superfund Health Risk Technical Support Center) Provisional Peer Reviewed Toxicity Values; and (3) other sources of information such as the California EPA's toxicity values and the Agency for Toxic Substances Disease Registry (ATSDR) minimal risk levels for non-carcinogenic compounds. This hierarchy was used by the U.S. EPA Region 9 to calculate the most recent version of PRGs (October 20, 2004); therefore, the PRG

table (U.S. EPA, 2004a) will be used as a reference for toxicity data for this assessment. For radio-nuclides, toxicity criteria are available in other sources, including Federal Guidance Report No. 13 (U.S. EPA, 1999) and the U.S. EPA's Health Effect Assessment Summary Tables (HEAST) (U.S. EPA, 1997e). Toxicity data for COPCs evaluated in the BRA are summarized on Table 6-24. Toxicity data for radionuclides evaluated in this risk assessment are summarized on Table 6-25.

6.4.6   Methodology for Risk Assessment of Lead

U.S. EPA (2001b) has determined that it is inappropriate to develop a RfD for inorganic lead compounds. In contrast to risk assessment techniques for most other chemicals, the toxic effects of lead usually are correlated with observed or predicted blood lead concentrations rather than with calculated intakes or doses. Consequently exposure to lead typically is evaluated using a pharmacokinetic model that incorporates specific assumptions for lead absorbtion from water, diet, and soil and that predicts lead levels in blood. Two such pharmacokinetic models were used to evaluate lead exposures for IR Site 2: the DTSC's LeadSpread 7 Model and the U.S. EPA's Adult Lead Model (ALM).

6.4.6.1   LeadSpread 7 Model

Risk assessment of lead was performed using the current version of the Lead Risk Assessment Spreadsheet, LeadSpread 7, developed by DTSC (1999b). The LeadSpread model calculates blood lead concentrations in adults (residents and workers) and children and compares the calculated concentration to a baseline of 10 µg/dl, which is generally accepted as the blood lead concentration below which toxic effects have not been observed. LeadSpread evaluates the following exposure pathways: ingestion of site soil/dust; dermal contact with site soil/dust; inhalation of site soil/dust; background air inhalation; ingestion of drinking water; ingestion of market basket food; and ingestion of home-grown produce (optional). For the ingestion route of exposure, blood lead (PbB) concentrations are calculated using proportionality constants (child and adult) between the amount of lead ingested per day and the PbB concentration. Similar proportionality constants are employed for the inhalation and dermal contact routes of exposure. All contributing sources to PbB are summed, and the summed value then is defined as the geometric mean. A fixed value for the geometric standard deviation is imposed (1.6) and various percentiles of the distribution of expected PbB levels for the overall PbB concentration are calculated.

LeadSpread can be used to model all of the receptor scenarios developed for the site. These receptors include a Park Ranger/Tour Guide, a Ranger/Habitat Restoration Supervisor, an Adult Site Visitor, a Child Site Visitor, and a Construction/Excavation Worker. The rationale for selection of these receptor scenarios was described previously in Section 6.4.1.

The input parameters used for modeling IR Site 2 in LeadSpread are shown in Table 6-26. Where possible, site-specific values were used for the input parameters. For the input parameter "Lead in Air", a value of 0.008 µg/m^3 was selected based on monitoring data available from the monitoring station nearest the site. In the California Air Resources Board (CARB) Annual Toxics Summary, the Fremont-Chapel Way was the monitoring station nearest the site that measured lead in air in recent years. This station reported that the 90th percentile of concentrations for 2002 was 0.008 µg/m^3 (CARB, 2005). This value was also the 90th percentile for 2001. The mean concentration reported in 2000 was 0.005 µg/m^3 (CARB, 2005). No mean was reported for 2001 or 2002; however, the minimum concentration was 0.0015 µg/m^3 and the maximum was 0.012 µg/m^3 in 2002. Therefore, a concentration of 0.008 µg/m^3 was selected as a conservative estimate of lead in air at the site.

For the input parameter "Lead in Soil/Dust", the 95% upper confidence limit of the mean was used, as recommended by DTSC. The value used in each model run varies between the landfill and wetland and with the soil interval to which the particular receptor was exposed. Surface soil is defined as the 0 to 1 ft

depth interval and surface and subsurface soil combined is being defined as the 0 to 10 ft depth interval (or depth to groundwater, which ever comes first). The concentrations of lead in soil are presented in Table 6-27. The Park Ranger/Tour Guide, Adult Site Visitor, and Child Site Visitor are assumed to be exposed to surface soil (0 to 1-ft depth) only. The Ranger/Habitat Restoration Supervisor and the Construction/Excavation Worker are assumed to be exposed to the surface and subsurface soil combined (0 to 10-ft or depth to groundwater).

Lead in water was assumed to be the default value of 15 µg/L recommended by DTSC. This value is the MCL for lead in drinking water.

Because all receptor scenarios for this site are non-residential, ingestion of homegrown produce was not included in the assessment (i.e., a value of 0% was selected for homegrown produce grown at the site and consumed by the receptors).

The concentration of respirable dust in air was assumed to be 20 µg/m^3 based on results from the nearest monitoring stations. The respirable dust concentration is used along with the lead in soil/dust concentration in the inhalation pathway to calculate the inhalation of lead attributable to the soil at the site. The Annual Bay Area Air Quality Summary for 2004 reports the average annual PM$_{10}$ concentrations for the San Pablo station as 21.2 µg/m^3 and the Fremont station as 18.6 µg/m^3 (Bay Area Air Quality Management District [BAAQMD], 2005). The concentration of 20 µg/m^3 is approximately the average of the values for the two monitoring stations. This concentration is higher than the default value of 1.5 µg/m^3.

In addition to the input parameters, several exposure parameters were modified from the defaults to better reflect the exposure scenarios developed for IR Site 2. Parameters modified included the number of days per week, skin area, and soil ingestion rates. The modified values for each receptor are shown in Table 6-28. Other exposure parameters remained at the default values developed by DTSC. These default values are shown in the spreadsheets of the results presented in Table J-9 in Appendix J.

6.4.6.2 Adult Lead Methodology

For purposes of comparison to LeadSpread, the U.S. EPA's ALM (U.S. EPA, 2003b and 2001c) also was used to calculate PbB levels for all receptor scenarios except the Child Site Visitor. The ALM is based on calculation of the PbB levels for the fetus of a pregnant female worker exposed to lead contaminated soils and dust in a non-residential setting. The model also calculates the maternal PbB level; however, PRGs calculated from the ALM are based on the fetal PbB concentration with a target of 10 µg/dl. The methodology uses a simplified representation of lead biokinetics to predict quasi-steady state blood lead concentrations among adults who have relatively steady state patterns of site exposures. The default values for soil and dust exposure parameters are based on an indoor worker scenario.

For the ALM, U.S. EPA's Metals Workgroup (see http://www.epa.gov/superfund/programs/lead/products.htm) recommends using the updated values recommended by the NHANES III Study for the geometric standard deviation (GSD$_i$) and baseline blood lead levels (PbB$_0$). GSDi is a measure of the inter-individual variability in blood lead concentrations in a population whose members are exposed to the same nonresidential environmental lead levels. The PbB$_0$ is the typical blood lead concentration (µg/dl) in women of child-bearing age in the absence of exposures to the site being assessed. For site applications of the ALM, the NHANES III study (Thayer and Diamond, 2002) states that estimates of GSD$_i$ and background blood lead (PbB$_0$) could be based on either race/ethnicity or geographic categories determined to be appropriate based on the demographic or geographic characteristics of the site. Because the site is located in California and the receptors potentially could come from any race or ethnic background, the values for the Western Region for "All" race/ethnic groups were selected. The GSD$_i$ selected was 2.11 and the background PbB$_0$ was 1.4 µg/dl. Because the ALM assumes an indoor worker, the soil

ingestion exposure terms were modified to reflect those developed for the site-specific receptor scenarios. Also, the number of days of exposure per year were modified to reflect the site-specific receptor scenarios. The modified exposure terms for each of the receptors are summarized in Table 6-29. Other exposure parameter values not listed in this table were set at the ALM default values and are shown in the spreadsheets provided in Table J-10 in Appendix J.

6.4.6.3 Integrated Exposure Uptake Biokinetic Model for Lead in Children

The U.S. EPA's Integrated Exposure Uptake Biokinetic Model for Lead in Children (IEUBK model) was considered for use in modeling the child receptor as a comparison to the LeadSpread results; however, it was decided that this model is not appropriate for the Site Visitor receptor scenario. Although the IEUBK model is designed specifically for evaluating children, it was developed for residential settings and is not recommended for intermittent exposures such as those likely to be encountered under the planned future use at IR Site 2. Because LeadSpread provides an estimate for childhood exposure and because the lead exposure concentrations for this receptor were low (less than California's recommended residential PRG for lead of 150 mg/kg), LeadSpread was considered an adequate exposure model for the child receptor.

6.4.7 Results of Baseline Risk Assessment

6.4.7.1 Results of Lead Risk Assessment

The results and conclusions of the risk assessment of lead using LeadSpread modeling and ALM are presented in this section.

6.4.7.1.1 LeadSpread Modeling

The results of the LeadSpread 7 modeling are summarized in Table 6-30. The spreadsheets showing the results for each receptor are presented in Table J-9 in Appendix J. The results indicate that the 99th percentile predicted PbB concentrations are below the target PbB concentration of 10 µg/dl for all five receptors exposed to the wetland area and for the Park Ranger/Tour Guide, the Adult Site Visitor, and the Child Site Visitor exposed to the landfill area.

However, for the landfill area, the two receptors that are assumed to have contact with surface and subsurface soils combined have a predicted PbB above the target PbB of 10 µg/dl. The 99th percentile predicted PbB concentration for the Ranger/Habitat Restoration Supervisor at the landfill is 17.9 µg/dl and the 95th percentile PbB concentration is 13.0 µg/dl. The 99th percentile PbB concentration for the Construction/Excavation Worker at the landfill is 63.4 µg/dl and the 95th percentile PbB concentration for this receptor is 45.8 µg/dl.

The 95% UCL values for the surface soil (0 to 1 ft) at both the landfill and the wetland area and for the combined surface and subsurface soil (0 to 10 ft) at the wetland are all below the California DTSC's residential PRG of 150 mg/kg and U.S. EPA's recommended residential screening level of 400 mg/kg. Only the 95% UCL for combined surface and subsurface soil (0 to 10 ft or depth to groundwater) at the landfill exceeds these screening values.

It should be noted that for the landfill's combined surface and subsurface soil (0 to 10 ft depth) interval, the 95% UCL is driven largely by 3 of the 142 samples. Two samples have elevated lead concentrations— 62,200 mg/kg at a depth of 2-3 ft and 59,600 mg/kg at a depth of 7-8 ft. The third data point is 5,640 mg/kg at a depth of 5-6 ft. The remaining 139 data points are less than 1,000 mg/kg, and much lower than the 95% UCL of 4,789 mg/kg. Of these 139 samples, only seven exceed the U.S. EPA residential screening value of 400 mg/kg.

The LeadSpread model back calculates PRGs for each receptor. The PRGs are the soil concentrations estimated to result in the target blood lead level (10 μg/dl) for a given exposure scenario. Therefore, exposure to soil concentrations below the PRG should result in PbB below the target for a particular receptor. PRGs are calculated for both the 99th percentile estimate of PbB (PRG-99) and the 95th percentile estimate of PbB (PRG-95). Lead concentration contours were plotted for the PRGs (both PRG-99 and PRG-95) for both the ranger/restoration supervisor and the excavation/construction worker receptor scenarios in order to show the areas of the site where exposure to surface and subsurface soils combined could potentially result in PbB above the 10 μg/dl target in these receptors (Figure 6-8). As the figure shows, only limited portions of the landfill exceed the PRGs and no portions of the wetland area exceed the PRGs. Also, all samples exceeding the PRGs are located in the subsurface soils (>1 ft depth). Although possible, it seems unlikely that the receptors, particularly the ranger/restoration supervisor whose PRGs are much higher, would be exposed to those areas and layers that are above the PRGs for their entire exposure duration.

Because the two elevated lead concentrations cause the 95%UCL to be inflated, the LeadSpread model was run again for comparison using the average lead concentration in the landfill for surface and subsurface soils combined (1,015 mg/kg) for the two receptors potentially exposed to this soil interval. Results from LeadSpread using the average soil concentration are presented in Table 6-31. Using the average concentration, predicted PbB concentrations for the Ranger/Habitat Restoration Supervisor are below the target of 10 μg/dl and the PbB concentrations for the Construction/Excavation Worker are only slightly above 10 μg/dl. Also, for comparison, the Construction/Excavation Worker scenario was calculated using updated values for the geometric standard deviation (1.4 rather than the default of 1.6) and bioavailability (0.3 rather than the default of 0.44), as recommended in the Navy's Standard Operating Procedures for investigating lead risk (DeGrandchamp, 2005), which also recommends use of the average rather than the 95%UCL in LeadSpread. Using these updated values, the predicted PbB concentrations for the Construction/Excavation Worker are below the target of 10 μg/dl.

6.4.7.1.2   Adult Lead Methodology Results

Results of the ALM are similar to those from LeadSpread (Table 6-32). The ALM results indicate that exposure to concentrations in the wetland would not cause concern for adult or fetal blood lead levels in any of the receptor scenarios. Likewise, for the Park Ranger/Tour Guide and the Adult Site Visitor receptors at the landfill, PbB levels would not exceed the target of 10 μg/dl in either the adult or the fetus. However, for the Ranger/ Habitat Restoration Supervisor and the Construction/Excavation Worker at the landfill area, PbB concentrations would exceed the target of 10 μg/dl in both the adult and the fetus, when the 95% UCL value is used for the soil lead concentration. As discussed above, the 95% UCL for the landfill surface and subsurface soil interval combined is inflated largely due to two elevated lead concentrations detected in the landfill subsurface soils. Therefore, as was done for LeadSpread, the ALM also was run using the mean soil lead concentration for comparison. Results using the mean soil concentrations are presented in Table 6-33. Using the mean soil concentration, the adult PbB concentrations are below the 10 μg/dl target for both the Ranger/Habitat Restoration Supervisor receptor and the Construction/Excavation Worker. The fetal PbB levels calculated for the Ranger/Habitat Restoration Supervisor was 12.5 μg/dl, which only slightly exceeds the target, whereas the fetal PbB for the Construction/Excavation Worker was somewhat higher than the target at 21.9 μg/dl.

6.4.7.1.3   Conclusions

Both LeadSpread and the ALM predict PbB concentrations that exceed the target PbB of 10 μg/dl for the Ranger/Habitat Restoration Supervisor site worker and the Construction/Excavation Worker in the landfill area when the 95% UCL concentration for the combined surface and subsurface soil interval. Both receptor scenarios assume exposure to soils in the 0 to 10 ft depth (or depth to groundwater) range

and both receptors are assumed to have relatively high soil ingestion rates. As noted above, the sub-surface soil concentrations observed in the landfill include two elevated lead concentrations, whereas the majority data points are below U.S. EPA's recommended residential screening level of 400 mg/kg. As illustrated in Figure 6-8, very few isolated areas in the landfill have concentrations above LeadSpread's PRGs for the Construction/Excavation Worker (PRG-95 = 844 mg/kg; PRG-99 = 539 mg/kg) and for the Ranger/Habitat Restoration Supervisor (PRG-95 = 3,459 mg/kg; PRG-99 = 2,210 mg/kg). It seems unlikely that the future receptors would be exposed to concentrations exceeding the PRGs over the entire duration of their exposure, particularly the Ranger/Habitat Restoration Supervisor.

Using the average lead concentration for surface and subsurface combined in the landfill area, the results of LeadSpread predict PbB levels below the target for the Ranger/Habitat Restoration Supervisor. Use of the average lead concentration for the Construction/Excavation Worker predicts levels only slightly above the target PbB level if default values are used, and, if updated values are used for the geometric standard deviation (1.4) and bioavailability (0.30), the PbB levels are below the target. Using average lead concentration in the ALM, PbB concentrations for the adult worker are below the target PbB level; however, the fetal PbB concentration exceeds the target of 10 µg/dl.

Lead concentrations in the wetland area are not expected to produce PbB levels above the target of 10 µg/dl for any of the receptor scenarios based on modeling with LeadSpread or the ALM. Likewise, the Park Ranger/Tour Guide, the Adult Site Visitor, and Child Site Visitor receptors exposed to surface soils at the landfill are not expected to have PbB levels above the target.

6.4.7.2 Total Risks

This section presents and discusses the results of the baseline human health risk assessment for IR Site 2. Results are provided in three ways. First, tables are presented that summarize the total risk for each receptor along with with a subtotal risk for each exposure route (Tables 6-34 and 6-35). Second, tables identifying the risk driving constituents (i.e., an individual constituent that has a risk $>1 \times 10^{-6}$ or a non-cancer hazard >1) for each receptor are presented (Tables 6-36 and 6-37). Finally, tables are presented that summarize risk-driving constituents by exposure media in the landfill area and the wetland area, (Tables 6-38 and 6-39).

In addition to the summary tables presented in this section, tables that list the calculated risks for all individual chemicals for all receptors that were evaluated in the BRA are provided in Table J-11 through J-26. The total risk values shown on these tables represent the risk or hazard that each receptor would incur, under the stated exposure assumptions for that receptor; and, the risk values include contributions from constituents that are associated with past activities at IR Site 2, constituents that are naturally-occurring in the environment, and constituents that are present due to anthropogenic sources other than the Navy's site activities at IR Site 2 (e.g., SVOCs/PAHs in imported soil used as fill material and atmospheric deposition).

The calculated cancer risk and non-cancer hazard for each receptor is summarized on Tables 6-34 (Landfill Area) and 6-35 (Wetland Area). The following results were obtained for the landfill area (Table 6-34).

- All four receptors had a total cancer risk $>1 \times 10^{-6}$. Cancer risks range from 7.13×10^{-6} (Park Ranger/Restoration Supervisor) to 2.75×10^{-5} (Park Ranger/Tour Guide). Total risk is attributable to exposure via multiple exposure routes as shown on Table 6-34. For the Park Ranger/Tour Guide and the Site Visitor, three exposure routes pose a cancer risk $>1 \times 10^{-6}$: soil ingestion, soil dermal contact, and external radiation. For the Park Ranger/Restoration Supervisor and the Construction/Excavation Worker, the exposure

routes that pose a cancer risk >1 × 10⁻⁶ include soil ingestion, groundwater dermal contact, and external radiation. The Park Ranger/Restoration Supervisor and the Construction/Excavation Worker were assumed to be exposed to groundwater; whereas, the other two recptors were not.

- Two receptors had a total non-cancer hazard >1.0: the Park Ranger/Restoration Supervisor and the Construction/Excavation Worker. The total non-cancer hazard for these two receptors is 1.91 (Park Ranger/Restoration Supervisor) and 13.8 (Construction/Excavation Worker). No receptor had an individual exposure route that poses a non-cancer hazard >1 except the Construction/Excavation Worker. For this receptor, three exposure routes have a non-cancer hazard >1: soil ingestion, inhalation of VOCs in soil, and groundwater dermal contact.

The following results were obtained for the wetland area (Table 6-35).

- All four receptors had a total cancer risk >1 × 10⁻⁶. Cancer risks range from 7.8 × 10⁻⁶ (Site Visitor) to 2.36 × 10⁻⁵ (Construction/Excavation Worker). For the Park Ranger/Tour Guide and the Site Visitor, the exposure routes that have a cancer risk >1 × 10⁻⁶ include soil ingestion, soil dermal contact, and external radiation. For the Park Ranger/Restoration Supervisor, dermal contact with groundwater and dermal contact with surface water have a cancer risk >1 × 10⁻⁶. For the Construction/Excavation Worker, only groundwater dermal contact has a cancer risk >1 × 10⁻⁶.

- Two receptors had a total non-cancer hazard >1.0: the Park Ranger/Restoration Supervisor and the Construction/Excavation Worker. Total non-cancer hazard for these two receptors are 2.95 (Park Ranger/Restoration Supervisor) and 26.2 (Construction/Excavation Worker). Non-cancer hazard for the Park Ranger/Restoration Supervisor and the the Construction/Excavation Worker is driven by dermal contact with groundwater.

Tables 6-36 and 6-37 identify the risk driving constituents, by receptor, for the landfill area and the wetland area, respectively. As stated previously, a potential risk driver is a constituent that has a risk >1 × 10⁻⁶ or a non-cancer hazard >1. The following results were obtained for the landfill area (Table 6-36).

- Eight constituents were identified that have a cancer risk > 1 × 10⁻⁶ for an individual receptor/exposure route combination, including: arsenic, benzo(a)pyrene, benzo(k)fluoranthene, delta-HCH, total PCB, Ra-226, Ra-228, and TEQ. The total number of risk-driving constituents varies by receptor.

- Two constituents were identified that contribute a non-cancer hazard >1 for an individual receptor/exposure route combination, including: total PCBs and naphthalene. For the Park Ranger/Tour Guide and the Site Visitor, no single constituent has a non-cancer hazard >1 for any exposure route.

The following results were obtained for the wetland area (Table 6-37).

- Eight constituents were identified that have a cancer risk >1 × 10⁻⁶ for an individual receptor/exposure route combination, including: arsenic, benzo(a)pyrene, benzo(k)fluoranthene, dibenz[ah]anthracene, dieldrin, total PCBs, Ra-226, and Ra-228. The total number of risk driving constituents varies by receptor.

- Total PCB is the only constituent that contribute a non-cancer hazard >1 for an individual receptor/exposure route combination. For the Park Ranger/Tour Guide and the Site Visitor, no single constituent has a non-cancer hazard >1 for any exposure route.

Tables 6-38 and 6-39 identify the risk driving constituents for each exposure media in the landfill area and the wetland area, respectively, that was evaluated in the BRA. The following results were obtained for the landfill area (Table 6-38).

- Potential risk drivers in landfill surface soil include: arsenic, benzo(a)pyrene, benzo(k)fluoranthene, Ra-228, and total PCBs.

- Potential risk drivers in landfill surface/subsurface soil combined include: Ra-226, Ra-228, and naphthalene.

- Potential risk drivers in groundwater in the landfill area include: total PCBs, *delta*-HCH, and TEQ.

The following results were obtained for the wetland area (Table 6-39).

- Potential risk drivers in wetland surface soil include: arsenic and benzo(k)fluoranthene.

- Potential risk drivers in wetland surface/subsurface soil combined include: Ra-226 and Ra-228.

- Potential risk drivers in groundwater in the wetland area include: dieldrin and total PCBs.

- Potential risk drivers in surface water in the wetland area include: benzo(a)pyrene and dibenz[ah]anthracene.

6.4.7.3   Background Risks

As stated in Section 6.4.7.2, the total risk values represent the total risk or hazard that each receptor would incur, including contributions from constituents that are associated with past activities at IR Site 2, constituents that are naturally occurring in the environment, and constituents that are present due to anthropogenic sources other than past site activities performed by the Navy at IR Site 2. In this section, constituents that were identified as potential risk drivers in the previous section but that also either occur naturally or were detected in samples collected during the various reference sampling activities (i.e., Ra-226 background established by during radiological surveying [TTFW, 2005] and RI sampling at CCSP) are identified and discussed. In addition, corresponding risks associated with these background constituents is quantified and compared to total risks presented previously in order to provide context for the total risk values.

- Arsenic was identified as a potential risk driver in surface soil in both the landfill area and the wetland area for the Park Ranger/Tour Guide and the Site Visitor. However, arsenic is a naturally occurring constituent in soils at Alameda Point, having an average concentration of 9.4 mg/kg (TtEMI, 2001). The EPC that was calculated for arsenic for use in the risk assessment is 5.33 mg/kg for landfill surface soil and 8.46 mg/kg for wetland surface soil. These concentrations are below the background concentration for Alameda Point soils; therefore, this suggests that arsenic is probably not a site-related contaminant and thus should not be included in the risk assessment. Table 6-40 compares the calculated risks for arsenic for the Park Ranger/Tour Guide and the Site Visitor based on the EPCs in landfill and wetland surface soil to the background risk for arsenic based on the average arsenic concentration in Alameda Point soils. As shown, background risks exceed the total risks for arsenic.

Based on this, it is recommended that arsenic should not be considered a potential risk driver in landfill and wetland surface soil, as it appears that it is naturally present in soils at Alameda Point at concentrations similar to those measured at IR Site 2.

Some of the organic compounds that were identified as risk-drivers in the BRA for IR Site 2 were detected in the upland reference area soil samples collected at CCSP, suggesting a possible regional anthropogenic source for these constitutents. For example, benzo(a)pyrene, benzo(k)fluoranthene, and total PCBs were identified as potential risk drivers in surface soil in the landfill area (see Table 6-38). However, all of these constituents were also detected in one or more of the three upland reference area soil samples collected from CCSP. The small number of samples collected from CCSP precludes performing statistical analyses to determine if the concentrations of these constituents at IR Site 2 are significantly different (higher) than the concentrations detected at CCSP. However, the average concentration of one compound in the CCSP soils, benzo(a)pyrene, is higher than the 95% UCL concentration in the landfill surface soil at IR Site 2. This suggests the possibility that benzo(a)pyrene (and possibly other compounds) may be present in Site 2 soils because of a regional anthropogenic source, such as atmospheric deposition or emplacement of dredged sediments from San Francisco Bay, rather than previous activities at IR Site 2. The background risk for benzo(a)pyrene, based on the average concentration detected in the upland reference area at CCSP, is almost identical to the total risk that was calculated for the site (Table 6-41).

Based on this, it appears that it is possible that benzo(a)pyrene and possibly other organic constituents may be present in soil due to anthropogenic sources other than previous activities at IR Site 2. However, as stated previously, it is not possible to rule out IR Site 2 as the source of these constituents. Nevertheless, the possibility that a background source exists for these constituents should be taken into consideration when making risk management descisions about the need to remediate based on these constituents.

Ra-226 and Ra-228 were identified as potential risk drivers in surface soil (Ra-228 only) and combined surface/subsurface soil in the landfill area and in combined surface/subsurface soil in the wetland area for the Park Ranger/Tour Guide and Site Visitor (see Tables 6-38 and 6-39). However, for Ra-226, a background concentration of 0.365 pCi/g has been determined for Alameda soils (TTFW, 2005). Based on the background concentration, background risks shown are almost equal to the total risk posed by this constituent (Table 6-42). Thus, even if Ra-226 is a site-related contaminant, the levels that are present at the site do not appear to pose risk that is above background risk from this constituent. In addition, the presence of Ra-228 is likely to be the result of natural processes (i.e., the decay of thorium-232) rather than historic site activities.

Based on this, it is recommended that Ra-226 and Ra-228 should not be considered a potential risk driver in landfill and wetland surface soil.

6.4.7.4   Uncertainties in the Baseline Risk Assessment Results

Risk results presented in this report are best estimates based on the latest information and techniques available for predicting risk. Nevertheless, there is uncertainty associated with the results. The two primary sources of uncertainty are:

- Model uncertainty (i.e., methods/models used to calculate exposure and risk); and,
- Parameter uncertainty (i.e., uncertainty in model input parameters).

Model uncertainty is not discussed because standard, accepted exposure and risk models have been employed in this risk assessment; therefore, it is assumed that the formulations of the models used to predict exposure and risk are valid. Large uncertainties can often arise in risk estimates that are based on models that simulate the fate/transport of contaminants. However, for IR Site 2, risks are primarily based on measured contaminant data and there is little dependency on the use of fate/transport modeling to predict exposure point concentrations. One exception is the use of the VF and PEF, which are mass-transfer factors (i.e., models) used to predict contaminant concentrations in outdoor air from volatilization of VOCs in soil and from wind suspension of soil containing adsorbed contaminants, respectively. Both the VF and PEF are calculated using models developed by U.S. EPA for use in risk assessments; therefore, the uncertainty in these models is considered acceptable.

Parameters involved in the risk assessment can be categorized according to the step in the risk assessment in which they occur (i.e., hazard identification, exposure assessment, dose-response [toxicity] assessment, and risk characterization). The various parameter uncertainties and the likely impact of these uncertainties on the calculated risks for IR Site 2 are summarized in this section and on Table 6-43.

The major uncertainty associated with the hazard identification process is the identification of constituents of potential concern (i.e., were all COPCs identified?). The potential for failing to identify COPCs for the IR Site 2 risk assessment is relatively low because all environmental media evaluated in the risk assessment (i.e., soil, groundwater, surface water) were analyzed for a very broad suite of constituents; and, analytical methods capable of achieving low limits of detection were employed. In addition, all constituents that were detected in the various environmental media were included in the risk assessment; no constituents were "screened out" based on FOD, concentration, and whether or not they are naturally occurring in the environment. Because of these factors, the amount of uncertainty in the calculated risks resulting from uncertainty in the COPCs is considered low (Table 6-43).

Another aspect identifying COPCs that imparts uncertainty to the calculated risks is the situation that occurs with a constituent that has a FOD equal to zero but a MDL in one or more samples that exceeds a conservative health-based screening level (e.g., PRG). Because of the elevated MDL, it is not possible to conclude that such a constituent is not present at a level that could pose a risk; however, because the constituent was not detected, it is not possible to evaluate it quantitatively in the risk assessment. Consequently, the actual site risks could be slightly underestimated if these constituents are present at the site. Table 6-44 identifies constituents with a FOD equal to zero but with a MDL in one or more samples that exceeds a conservative health-based screening level.

As discussed in Section 5.1 of this RI Report, there is some uncertainty associated with the total PCB results estimated using the NOAA NS&T method related to apparent discrepancies between those results and total PCB results estimated using the sum of Aroclors. To assess the potential significance of this uncertainty, alternative risk calculations were conducted using total PCB values based on the sum of Aroclors. The total PCB results based on sum of Aroclors were recalculated in three different ways: as the sum of original Aroclors; as the sum of requantified Aroclors; and as the sum of Aroclors on a point-by-point basis factoring in the more conservative (i.e., higher) of the original or modified Aroclor results. To be conservative and assess the greatest possible impact of this source of uncertainty, the highest 95% UCL value of the three different Aroclor sums for the landfill and wetland were used to conduct an alternative risk calculation. Tables 6-45 and 6-46 present the chemical-specific and cumulative risks, respectively, based on total PCBs calculated using the NOAA NS&T method and the most conservative sum of Aroclors method. As shown, there were only marginal increases in cancer and non-cancer risks from PCBs when using the most conservative total PCB concentrations calculated from the sum of Aroclors as compared to the NS&T method. Similarly, cumulative cancer risk and non-cancer risk at the site are essentially the same or only slightly higher when using total PCBs based on the conservative sum of Aroclors as compared to total PCBs based on the NOAA NS&T method.

Several parameters associated with the exposure assessment have uncertainties associated with them that impart uncertainty to the calculated risks, including EPCs, potential receptors and exposure assumptions evaluated in the risk assessment, and site physical data. Each of these is discussed below and summarized on Table 6-43.

- Based on U.S. EPA risk assessment guidance, the UCL of the arithmetic mean is used as the EPC because it is a conservative estimate of the average site-wide concentration that a receptor would be exposed to. The UCL is a statistic and thus by nature is uncertain; however, to minimize the uncertainty in the EPCs, UCLs were calculated using several statistical methods and the most appropriate value was selected based on factors such as distribution of the raw data (e.g., normal, lognormal) and FOD (see Appendix G for a discussion of methods used to calculate UCLs). The amount of uncertainty in the calculated risks resulting from uncertainty in the EPCs is considered low.

- Because of the uncertainty regarding the future use of the site, there is uncertainty in the receptors that were evaluated in the risk assessment and their activities/habits. Four receptors were identified and evaluated in the risk assessment, including two future site workers (Park Ranger/Tour Guide and Ranger/Habitat Restoration Supervisor), a Site Visitor, and a Construction/Excavation Worker. To minimize uncertainty in the calculated risks for the Park Ranger/Tour Guide and the Ranger/Habitat Restoration Supervisor, exposure assumptions and parameters for these receptors were taken from the NWS Seal Beach BRA which included these receptors (Bechtel, 2001). Furthermore, the exposure parameters used in the NWS Seal Beach risk assessment were developed in conjunction with the U.S. Fish and Wildlife Service, which was slated to receive the property from the Navy. Exposure assumptions and parameters for the Site Visitor and the Construction/ Excavation Worker were based on professional judgment and standard default exposure values recommended by U.S. EPA and other regulatory agencies. The uncertainty in the calculated risks is considered moderate based on these uncertainties; however, risks are more likely to be overestimated than underestimated because of the conservative nature of the exposure assumptions.

- A number of site parameters are required to calculate the VF and PEF terms for evaluating inhalation exposures, including properties of the site soils (bulk density, porosity, and moisture content) and properties of the site (e.g., area of the site, vegetative cover fraction, and wind speed). Whenever possible, site-specific data were used to describe these parameters; nevertheless, because a limited number of measurements were made for these parameters, there is some uncertainty in the reported values which imparts uncertainty in the calculated risks. When site-specific measurements were not made (e.g., wind speed), data from a location near the site or conservative assumptions were used; therefore, inhalation risks are more likely to be overestimated than underestimated. The amount of uncertainty in the calculated inhalation risks resulting from uncertainty in the VF and PEF is considered moderate.

- A special PEF that simulates wind suspension from vehicular traffic is used to calculate inhalation exposures for the Construction/Excavation Worker. Calculation of the PEF for this scenario requires information about the magnitude and duration of the future construction activities at the site (e.g., number and weight of vehicles traversing the site per day, distance traversed per day). However, it is not certain that construction will occur on site in the future; therefore, a hypothetical construction scenario was developed and evaluated. Therefore, the uncertainty in the calculated inhalation risks for the Construction/Excavation Worker are considered high. Even if construction does occur,

the inhalation risks for the Construction/Excavation Worker are probably overestimated because conservative assumptions were used to describe the hypothetical construction scenario.

As discussed below and summarized on Table 6-43, the primary aspects of the toxicity assessment that impart uncertainty to the calculated risks include uncertainty in the toxicity data and a lack of toxicity data for constituents detected at the site.

- Toxicity values are typically based on results of tests performed on animals and extrapolated to humans. However, because the most conservative values available are typically used, risks are more likely to be overestimated than underestimated.

- Subchronic toxicity data are not available for most constituents evaluated in the risk assessment; therefore, chronic toxicity data were used to calculate risks for the Construction/Excavation Worker even though the exposure duration is considered subchronic. This results in an overestimation of risks for the Construction/ Excavation Worker.

- Some constituents that were detected at the site could not be quantitatively evaluated in the risk assessment because toxicity data is not available for the constituents. Of the 192 constituents analyzed in soil, groundwater or surface water at IR Site 2, only 4 constituents could not be evaluated because of a lack of toxicity data (2,4,5,6-tetrachloro-*m*-xylene, bromochloromethane, magnesium, and sodium). If any of these constituents are present in the exposure media at the site, risks will be underestimated because contributions from these constituents are not included. Of these, bromochloromethane was never detected in any media at the site; and, two of the constituents are essential nutrients (magnesium and sodium). Therefore, the amount of uncertainty in the calculated risks resulting from missing toxicity data is low.

Finally, uncertainty in the calculated risks can arise from uncertainty in the way in which risks were calculated or aggregated, as discussed below and on Table 6-43.

- Because the calculated non-hazard risks (i.e., hazard indices) are relatively low (<10), they were not segregated according to effect and mechanism of action. Typically, segregation results in a reduction in the maximum non-cancer risk because effects are distributed rather than assuming that all non-carcinogenic constituents elicit the same effect, although some constituents can cause synergistic effects. In general, by not segregating non-cancer risks, it is more likely that the calculated risks are overestimated than underestimated.

- As stated previously, all detected constituents with toxicity data were evaluated in the risk assessment. Consequently, a portion of the calculated risks for each receptor is due to the presence of naturally occurring constituents or constituents that are present at the site because of regional anthropogenic sources (i.e., sources other than IR Site 2). The effect of including background and ambient constituents in the risk assessment is that the calculated risks overestimate the true risk that is due to chemical releases from IR Site 2 (i.e., incremental risks).

7.0 ECOLOGICAL RISK ASSESSMENT

7.1 ERA Objectives and Approach

This Ecological Risk Assessment (ERA) has been prepared in support of the RI evaluation at IR Site 2. The purpose of this ERA is to support the completion of RI activities, and to support recommendations for consideration in the FS process. Three primary and distinct habitat types have been identified at IR Site 2: upland/terrestrial (U), wetland (W), and wetland ponds (P). These habitats support a number of common receptors, as well as receptors that are unique to specific habitat types. The primary objectives of this ERA are to:

1. Conduct an assessment of ecological risk associated with the three general habitat types (i.e., upland, wetland, and wetland pond) at IR Site 2.

2. Identify key uncertainties that may impact the utility of the results of this ERA in a risk management context.

3. Provide a technical and scientific basis to identify potential ecological risk drivers that should be considered in order to develop a potential remedial footprint for consideration in the FS.

The approach used to conduct this ERA is consistent with the U.S. EPA's 1997 *Ecological Risk Assessment Guidance for Superfund* (U.S. EPA, 1997d), and the *Navy Policy for Conducting Ecological Risk Assessments* (CNO, 1999). As outlined in those guidance documents, a tiered process that encompasses eight steps is generally followed (see Figure 7-1). This eight step process is intended to be iterative, which is the same process that has been followed for this risk assessment.

A number of previous investigations have been conducted at IR Site 2 prior to 2005 and are summarized in Section 3.0 of this RI Report. The problem formulation step included in this ERA was developed based on the historical information collected during the past investigations, and was presented in the *Final RI Sampling Work Plan* (Battelle et al., 2005). To develop the RI Sampling Work Plan, Steps 1 and 2 (Screening-Level Assessment), Step 3 (Problem Formulation), Step 4 (Study Design and DQO Process), and Step 5 (Verification of Field Sampling Design) of the ERA process were followed. Although some data collected prior to 2005 were used to conduct quantitative risk analyses included in this report, the primary data set used to complete this ERA consists of data that were collected during RI sampling activities described in Section 4.0 and Appendix B, as part of the site investigation component of Step 6 (Site Investigation and Data Analysis).

This ERA presents the problem formulation component of the ERA that was originally presented in the RI Sampling Work Plan along with some updates (Section 7.2) resulting from RI sampling activities, and it includes a quantitative screening-level assessment of risk. The objective of the screening-level (or Tier 1) assessment is to separate areas, species, or contaminants at the site into two categories: those that have de minimus risk and can be eliminated from further assessment; and those where risk cannot be ruled out and may require further evaluation. In the screening-level assessment, conservative estimates and assumptions are used so that potential risk is actually overestimated. Thus, if the screening concludes that negligible risk exists, then there is strong support for a no further action recommendation. However, an indication of potential risk in the screening step does not necessarily mean that there is risk associated with an area, species, or contaminant. Rather, this outcome suggests that further evaluation may be justified before a decision can be made.

Compounds, habitats, and/or receptors that are determined to require further evaluation based on the results of the screening-level assessment are addressed in the baseline (or Tier 2) assessment. The first

step of the baseline assessment is to formally refine the problem formulation component of the risk assessment in order to focus the scope of the assessment on issues warranting additional evaluation. The baseline assessment also incorporates more reasonable assumptions of exposure based on site use, and includes additional site-specific data such as site-specific bioassay tests. Step 7 of the ERA process, risk characterization, consists of a presentation of baseline risk results that are organized according to assessment endpoint.

The ERA portion of this RI Report is organized into the following sections:

- Section 7.1, ERA Objectives and Approach. This section provides an introduction and overview of the ERA, including the objectives of and the approach to conducting the ERA.

- Section 7.2, Problem Formulation. This section describes the problem formulation phase, including the background and ecological setting, the selection of chemicals of potential ecological concern, the conceptual site model (CSM), assessment and measurement endpoints, and receptors of concern.

- Section 7.3, Screening-Level Assessment. This section presents the components of the screening-level assessment, including the exposure and effects assessments, and the presentation of screening-level results.

- Section 7.4, Baseline Assessment. This section presents the components of the baseline assessment, including refinement of problem formulation, baseline exposure and effects assessments, and risk characterization (including risk estimates, uncertainty assessment, and conclusions and recommendations).

The following appendices contain supplemental information used to prepare the ERA:

- Appendix C, Technical Summary of Toxicity and Bioaccumulation Testing for IR Site 2; and,

- Appendix K, ERA Exposure Point Concentrations and Detailed Food Web Model Output for Individual Receptors (including background/ambient exposure assessments).

7.2 Problem Formulation

The problem formulation is a formal process to develop and evaluate preliminary hypotheses concerning the likelihood and causes of ecological effects that may have occurred, or may occur, from human activities (U.S. EPA, 1998). It is conducted in the first step of the 8-step U.S. EPA ERA process as part of the initial screening-level evaluation. The results of the screening assessment are used to refine the problem formulation as the first step of the baseline ecological risk assessment (BERA) presented in Section 7.4. The problem formulation for IR Site 2 includes:

- Overview of the site setting;
- Evaluation of the physical and environmental setting;
- Development of the CSM and exposure pathway analysis;
- Selection of assessment and measurement endpoints (AEs and MEs);
- Selection of representative receptors; and,
- Summary of data quality assessment.

These elements constitute the overall study design for the ERA, and were developed to support (and were initially presented in) the *Final RI Sampling Work Plan* (Battelle et al., 2005).

7.2.1 Site Setting

As described in Section 2 of this RI Report, IR Site 2 consists of the West Beach Landfill ("landfill") which covers approximately 77 acres, and the West Beach Wetlands ("wetlands") which cover approximately 33 acres including two wetland ponds. The site borders San Francisco Bay and is located along the Pacific Flyway (Figure 2-1). For mammals and other land-based receptors, IR Site 2 is considered ecologically isolated due to its remote location in the southwestern corner of Alameda Point and the presence of a 6-ft-tall cyclone perimeter fence that prevents receptors from freely moving in and out of the site.

The physical and ecological characteristics of IR Site 2 were described in detail in Section 2 of this RI Report. Brief summaries of these site specifics are provided below; however, Section 2 contains a more detailed summary of the physical and ecological setting at IR Site 2.

7.2.1.1 Physical Setting

IR Site 2 is a relatively flat piece of land contained by berms and a seawall at the southwestern-most tip of Alameda Point. The area was created in 1956 with dredge fill materials and then used for waste disposal until 1978. The landfill area is estimated to encompass 77 acres and is referred to in this ERA as the "upland" area (Figure 2-2). The "wetland" area is used to describe those portions of the site characterized as periodically saturated, but not to a degree that results in significant long-term inundation or surface ponding (i.e., open water habitat). The topography and hydrology of the wetland areas at IR Site 2 are seasonally variable. Although a large 30- to 35-acre area within the southwestern portion of the site is generally considered wetlands (PRC, 1996; and Foster Wheeler, 2003), the actual topography and hydrology within this area is variable by the season. The open water resources in the wetlands consist of two perennial wetland ponds known as North Pond and South Pond (see Figure 2-2). For a detailed description of the site-wide physical attributes of the landfill, wetland, and wetland pond areas and the associated geology and hydrogeology, please refer to Section 2. These physical characteristics are important in identifying the chemical exposure pathways that may lead to ecological risk at IR Site 2.

7.2.1.2 Ecological Setting

Results from biological surveys, field observations, and natural history information (see Table 2-1) were reviewed to evaluate the potential ecological receptors present at IR Site 2 that may in some way be exposed to site-related chemicals. As defined above, the dominant habitats at IR Site 2 are upland (U), wetland (W), and wetland ponds (P).

Each of these habitats is summarized in detail in Section 2.10 of this RI Report along with the various plants, birds, mammals, reptiles, and amphibians potentially associated with the three habitat types. A brief summary of the ecological setting of each of these habitats that is present at IR Site 2 is provided below; further information is provided in Section 2.10.

7.2.1.2.1 Upland

The terrestrial environment of the 77-acre upland area is characterized as coastal prairie and northern coastal scrub. With the exception of a few isolated depressions, these upland terrestrial areas are never or rarely inundated. The majority of the upland area is highly disturbed from historical landfill activities. In the past, fill materials dredged from the wetlands and imported from other areas around Alameda Point

were used to partially cover the former landfill area. Plants, soil invertebrates, birds, and mammals that are identified as ecological receptors present in the upland areas and potentially exposed to site-related contaminants of potential ecological concern (COPECs) are described in Section 2.10.

7.2.1.2.2 Wetland

The wetland area in the southwestern corner of IR Site 2 is roughly a 33-acre area, fluctuating with variable topography and hydrology. Two wetland delineation programs have been conducted in the past (HRG, 1993a; TTWF, 2004) and identified those areas that were designated as jurisdictional wetland based on evaluation of soil types, hydrological characteristics, and vegetation types. For a detailed summary of the wetland delineation activities and a listing of plants, soil invertebrates, birds, and mammals that are identified as ecological receptors present in the wetland area of the site that could potentially be exposed to site-related COPECs, refer to Section 2.10.

7.2.1.2.3 Wetland Ponds

The physical characteristics of the wetland ponds vary considerably over the year. Fluctuations in salinity, dissolved oxygen, and turbidity make these ponds extremely variable in terms of aquatic habitat type and quality, making the ponds less hospitable for a wide variety of ecological receptors. Vegetation associated with ponds is present along the borders, but not within the open water areas. Existing data indicate that the diversity and abundance of the benthic and aquatic invertebrate community is relatively low. The low invertebrate diversity and abundance could be related to predation, seasonal drought conditions, or some combination of such factors. No mammals have been identified at the site that would preferentially use open water wetland pond habitat for foraging. However, the raccoon is an opportunistic omnivore and may eat aquatic species such as insects, frogs, and crayfish. The raccoon also may forage general wetland habitats and will consume virtually any plant or animal matter (U.S. EPA, 1993). Plants, benthic/aquatic invertebrates, fish, and birds that are identified as ecological receptors present in the wetland ponds of the site and could potentially be exposed to site-related COPECs are described in Section 2.10.

7.2.1.2.4 Special-Status Species Summary

No special-status mammalian species have been identified at IR Site 2. Special-status bird species have been observed at the Alameda Point, and may be supported by habitats existing at the site. Table 2-3 identifies California species of special concern, federally endangered, federally threatened, and California endangered species observed at former NAS Alameda. Special-status species to be considered in the ERA study design include the Canada goose (*Branta canadensis*), the California horned lark (*Erimophilsa aplestris actia*), the double-crested cormorant, the burrowing owl, and the northern harrier (*Circus cyanus*), and the Alameda song sparrow (*Melospiza melodia pusillula*).

7.2.2 Ecological CSM and Exposure Pathway Analysis

The CSM considers the ecological components of the site setting and identifies important site sources, release mechanisms, exposure media, exposure routes and receptors. The CSM schematically presents the relationship between potentially contaminated environmental media and classes of receptors at the site, and identifies potentially complete and significant pathways through which ecological receptors may be exposed to the identified environmental media. Current CSMs for the site have been developed based on historical and current site activities and on observations and biological surveys historically conducted at the site. For the purposes of the ERA, it is assumed that all of the site area will remain undeveloped and will provide potentially viable ecological habitat.

Simplified CSMs for the different habitat types at IR Site 2 have been developed and are presented as Figures 7-2, 7-3, and 7-4. Potential site contaminant sources, release mechanisms, exposure media, exposure routes, and receptors are described below.

7.2.2.1   Upland

Potential sources of chemicals in the upland area include the wastes disposed at the site from 1956 to 1978. As discussed in Section 2.11, wastes disposed at IR Site 2 include general municipal garbage, waste chemical drums, solvents, oily waste and sludge, paint waste, plating wastes, industrial strippers and cleaners, acids, mercury, PCB-containing liquids, batteries, low-level radiological waste from radium dials, scrap metal, asbestos, pesticides, tear gas, biological waste, creosote, dredge spoils, and waste medicines and reagents (Figure 2-10).

As illustrated in Figure 7-2, the primary release mechanisms for the landfill waste buried in the upland habitat at IR Site 2 include erosion of the soil cover, volatilization, and leaching into shallow ground-water. Media potentially exposed include surface and subsurface soils, soil gas, groundwater, and biota living within the surface soils.

Ecological guilds (i.e., groups of species that use the same set of resources in a similar manner) were identified to assist in determining if complete exposure pathways exist between site-related chemicals and ecological receptors. The ecological receptors potentially present in the terrestrial habitat of the upland area include plants, soil invertebrates, birds and mammals, and burrowing animals. Only those potentially complete exposure pathways likely to contribute significantly to total exposure were quantitatively evaluated in the ERA. Incomplete exposure pathways or those that result in insignificant exposure were not quantitatively evaluated in this assessment. Pathways that were not evaluated include exposure via dermal contact for wildlife and foliar uptake via plants. Dermal contact was considered a minor exposure pathway since research shows that dense coats prevent significant absorption of COPECs by wildlife (Peterle, 1991). Foliar uptake of ambient air also was considered a minor exposure pathway. Foliar uptake from soil is not addressed separately, but plant tissue data were collected and analyzed for COPECs to conduct foodweb models, thus foliar uptake of soil was captured in the risk assessment. Potentially complete pathways and associated ecological guilds for upland habitat receptors that were evaluated in this ERA include:

- Terrestrial plants – root contact with soil

- Soil invertebrates – direct contact including incidental ingestion of soil

- Birds – ingestion of soil and impacted plant or prey tissue
    - omnivorous birds
    - carnivorous birds
    - burrowing birds (inhalation pathway considered)

- Mammals – ingestion of soil and impacted plant or prey tissue
    - herbivorous mammals
    - omnivorous mammals
    - burrowing mammals
    - carnivorous mammals
    - burrowing mammals (inhalation pathway considered).

7.2.2.2 Wetland

Potential sources of chemicals in the wetland area include the historic spreading of dredge spoils from the Oakland Inner Harbor (IR Site 20), Pier Area (IR Site 24), the turning basin of Alameda Point, and Seaplane Lagoon (IR Site 17). Constituents potentially present in these sediments include metals, PCBs, SVOCs/PAHs, and/or pesticides. Additionally, waste-related chemicals from disposal practices in the landfill area also may have potentially impacted the wetland area via surface soil erosion. Runoff, wind suspension, overland transport, deposition to surrounding areas or groundwater discharge may release chemicals to the wetland surface soils, subsurface soils, or biota.

Ecological guilds were identified to assist in determining if complete exposure pathways exist between site-related chemicals and ecological receptors in the wetland area. The ecological receptors potentially present in the wetland habitat include plants, invertebrates, birds, and mammals. Only those potentially complete exposure pathways likely to contribute significantly to total exposure were quantitatively evaluated in the ERA. Incomplete exposure pathways or those that result in insignificant exposure were not quantitatively evaluated in this assessment. Potentially complete pathways for wetland receptors that were quantitatively evaluated in this ERA include:

- Wetland plants – root contact with soil

- Wetland invertebrates – direct contact including incidental ingestion of soil

- Birds – ingestion of soil and impacted plant or prey tissue

  - omnivorous birds
  - benthic-feeding birds
  - carnivorous birds
  - burrowing birds (inhalation pathway considered)

- Mammals – ingestion of soil and impacted plant or prey tissue

  - herbivorous mammals
  - omnivorous mammals
  - burrowing mammals
  - carnivorous mammals
  - burrowing mammals (inhalation pathway considered).

7.2.2.3 Wetland Ponds

Potential sources of chemicals in the wetland ponds are similar to those impacting the wetlands (such as those associated with dredge spoils) and the waste disposal practices in the adjacent landfill area. Chemicals in the wetland area may be present in pond surface sediment, surface water, or biota as a result of runoff, wind suspension, overland transport, and deposition to surrounding areas or lateral/vertical migration of groundwater associated constituents.

Ecological guilds were identified to assist in determining if complete exposure pathways exist between site-related chemicals and ecological receptors in the wetland ponds. The ecological receptors potentially present within the wetland ponds habitat include invertebrates, fish, and birds and mammals. Only those potentially complete exposure pathways likely to contribute significantly to total exposure were quantitatively evaluated in the ERA. Incomplete exposure pathways or those that result in insignificant exposure were not quantitatively evaluated in this assessment. Only those potentially complete exposure pathways

likely to contribute significantly to total exposure were quantitatively evaluated. Potentially complete pathways for wetland pond receptors that were quantitatively evaluated in the ERA include:

- Benthic invertebrates – direct contact including incidental ingestion of sediments
- Water column invertebrates and fish - direct contact including incidental ingestion of surface water
- Birds – ingestion of soil and impacted plant or prey tissue
    - herbivorous birds
    - benthic-feeding birds
    - fish-eating birds
- Mammals – ingestion of soil and impacted plant or prey tissue
    - omnivorous mammals.

7.2.3   Assessment and Measurement Endpoints

Assessment endpoints (AEs) are defined to represent each of the ecological guilds (i.e., groups of species that use the same set of resources in a similar manner) selected for inclusion in the ERA. Based on the AEs, specific risk questions (RQs) have been identified. The RQs reflect the selected AEs for the ERA in a question format, and are organized by the three major habitat types (i.e., upland, wetland, and wetland ponds). Measurement endpoints (MEs) also were selected to support the evaluation of AEs.

The AEs and RQs are intended to address resource protection at the population level. However, special-status species (e.g., protected species) should be evaluated with protection of the individual being para-mount. Receptor classes that are currently considered to include a special-status member are herbivorous birds, omnivorous birds, fish-eating birds, burrowing birds, and carnivorous birds.

With the exception of site-specific bioassay data, the same AEs and associated RQs and MEs are evalu-ated in both the screening-level and the baseline assessment. The screening-level assessment, however, incorporates more conservative exposure assumptions and toxicological thresholds, and the baseline assessment includes consideration of background or ambient exposure (whereas the screening-level assessment does not). These differences are discussed in detail in the exposure and effects assessment sections of the screening-level and baseline assessments included in this ERA. The following sections present a summary of the AEs, RQs, and MEs addressed in the ERA. Details regarding selection of representative receptors for each ecological guild represented by the AEs are presented in Section 7.2.4.

7.2.3.1   Upland

The following AEs, RQs, and associated MEs were identified for receptors present in the upland habitat (U), which is contained within the landfill area of IR Site 2.

U-AE1: Survival, growth, and reproduction of upland plants.

U-RQ1: Do COPEC concentrations in surface and subsurface soil pose unacceptable incremental ecological risk to upland plants?

ME1- Compare surface and subsurface soil chemistry data collected in upland area to phytotoxicity benchmarks.
ME2- Compare surface and subsurface soil chemistry data collected in upland area to Alameda Point reference soil values.

U-AE2: Survival, growth, and reproduction of upland soil invertebrates.

U-RQ2: Do COPEC concentrations in surface soil pose unacceptable incremental ecological risk to upland soil invertebrates?

> **ME1-** Compare site-specific surface soil chemistry data collected in upland area to soil invertebrate soil benchmarks.
> **ME2-** Compare site-specific surface soil chemistry data collected in upland area to Alameda Point reference soil values.

U-AE3: Survival, growth, and reproduction of omnivorous birds[2].

U-RQ3: Do COPEC concentrations in surface soil and associated biota pose unacceptable incremental ecological risk to omnivorous birds?

> **ME1-** Compare modeled dietary site and reference COPEC exposure to toxicity reference values (calculate hazard quotients).

U-AE4: Survival, growth, and reproduction of burrowing birds[2].

U-RQ4: Do COPEC concentrations in surface and subsurface soil and associated biota pose unacceptable incremental ecological risk to burrowing birds?

> **ME1-** Compare modeled dietary site and reference COPEC exposure to toxicity reference values (calculate hazard quotients).
> **ME2-** Compare measured site COPEC exposure to toxicity reference values (calculate hazard quotients).

U-AE5: Survival, growth, and reproduction of carnivorous birds[2 above].

U-RQ5: Do COPEC concentrations in surface soil and associated biota pose unacceptable incremental ecological risk to carnivorous birds?

> **ME1-** Compare modeled dietary site and reference COPEC exposure to toxicity reference values (calculate hazard quotients).

U-AE6: Survival, growth, and reproduction of herbivorous mammals.

U-RQ6: Do COPEC concentrations in surface soil and associated biota pose unacceptable incremental ecological risk to herbivorous mammals?

> **ME1-** Compare modeled dietary site and reference COPEC exposure to toxicity reference values (calculate hazard quotients).

U-AE7: Survival, growth, and reproduction of small omnivorous mammals.

U-RQ7: Do COPEC concentrations in surface soil and associated biota pose unacceptable incremental ecological risk to small omnivorous mammals?

> **ME1-** Compare modeled dietary site and reference COPEC exposure to toxicity reference values (calculate hazard quotients).

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[2] This ecological guild contains a special-status species. The selected representative receptor is either a special-status species, or conservatively represents exposure to a special-status species. Risk results for these endpoints will be interpreted with the understanding that protection must occur at the level of the individual.

U-AE8: Survival, growth, and reproduction of burrowing mammals.

U-RQ8: Do COPEC concentrations in surface and subsurface soil and associated biota pose unacceptable incremental ecological risk to burrowing mammals?

> **ME1-** Compare modeled dietary site and reference COPEC exposure to toxicity reference values (calculate hazard quotients).
> **ME2-** Compare measured site COPEC exposure to toxicity reference values (calculate hazard quotients).

U-AE9: Survival, growth, and reproduction of carnivorous mammals.

U-RQ9: Do COPEC concentrations in surface soil and associated biota pose unacceptable incremental ecological risk to carnivorous mammals?

> **ME1-** Compare modeled dietary site and reference COPEC exposure to toxicity reference values (calculate hazard quotients).

7.2.3.2   Wetland

The following AEs, RQs, and associated MEs were identified for receptors present in the wetland (W) habitat of IR Site 2.

W-AE1: Survival, growth, and reproduction of wetland plants.

W-RQ1: Do COPEC concentrations in soils pose unacceptable incremental ecological risk to wetland plants?

> **ME1-** Compare surface and subsurface soil chemistry data collected from the wetland area to phytotoxicity benchmarks.
> **ME2-** Compare surface and subsurface soil chemistry data collected from the wetland area to CCSP wetland soil reference values.

W-AE2: Survival, growth, and reproduction of terrestrial invertebrates.

W-RQ2: Do COPEC concentrations in soils pose unacceptable incremental ecological risk to terrestrial invertebrates?

> **ME1-** Compare site-specific surface soil chemistry data collected from the wetland area to soil invertebrate soil benchmarks.
> **ME2-** Compare site-specific surface soil chemistry data collected from the wetland area to CCSP wetland soil reference values.

W-AE3: Survival, growth, and reproduction of omnivorous birds[2 above (p. 7-8)].

W-RQ3: Do COPEC concentrations in surface soil and associated biota pose unacceptable incremental ecological risk to omnivorous birds?

> **ME1-** Compare modeled dietary site and reference COPEC exposure to toxicity reference values (calculate hazard quotients).

W-AE4: Survival, growth, and reproduction of benthic-feeding birds.

W-RQ4: Do COPEC concentrations in surface soils, sediments, and associated biota pose unacceptable incremental ecological risk to benthic-feeding birds?

ME1- Compare modeled dietary site and reference COPEC exposure to toxicity reference values (calculate hazard quotients).

W-AE5: Survival, growth, and reproduction of burrowing birds[2 above (p. 7-8)].

W-RQ5: Do COPEC concentrations in surface and subsurface soil and associated biota pose unacceptable incremental ecological risk to burrowing birds?

ME1- Compare modeled dietary site and reference COPEC exposure to toxicity reference values (calculate hazard quotients).
ME2- Compare measured site COPEC exposure to toxicity reference values (calculate hazard quotients).

W-AE6: Survival, growth, and reproduction of carnivorous birds[2 above (p. 7-8)].

W-RQ6: Do COPEC concentrations in surface soil and associated biota pose unacceptable incremental ecological risk to carnivorous birds?

ME1- Compare modeled dietary site and reference COPEC exposure to toxicity reference values (calculate hazard quotients).

W-AE7: Survival, growth, and reproduction of herbivorous mammals.

W-RQ7: Do COPEC concentrations in surface and subsurface soil and associated biota pose unacceptable incremental ecological risk to herbivorous mammals?

ME1- Compare modeled dietary site and reference COPEC exposure to toxicity reference values (calculate hazard quotients).

W-AE8: Survival, growth, and reproduction of omnivorous mammals.

W-RQ8: Do COPEC concentrations in surface and subsurface soil and associated biota pose unacceptable incremental ecological risk to omnivorous mammals?

ME1- Compare modeled dietary site and reference COPEC exposure to toxicity reference values (calculate hazard quotients).

W-AE9: Survival, growth, and reproduction of carnivorous mammals.

W-RQ9: Do COPEC concentrations in surface and subsurface soil and associated biota pose unacceptable incremental ecological risk to carnivorous mammals?

ME1- Compare modeled dietary site and reference COPEC exposure to toxicity reference values (calculate hazard quotients).

W-AE10: Survival, growth, and reproduction of burrowing mammals.

W-RQ10: Do COPEC concentrations in surface and subsurface soil and associated biota pose unacceptable incremental ecological risk to burrowing mammals?

ME1- Compare modeled dietary site and reference COPEC exposure to toxicity reference values (calculate hazard quotients).
ME2- Compare measured site COPEC exposure to toxicity reference values (calculate hazard quotients).

7.2.3.3　Wetland Ponds

The following AEs, RQs, and associated MEs were identified for receptors present in the wetland ponds (P) habitat of IR Site 2.

P-AE1: Survival, growth, and reproduction of water column invertebrates.

P-RQ1: Do COPEC concentrations in the surface water of the wetland ponds pose unacceptable incremental ecological risk to water column invertebrates?

> **ME1**- Compare site-specific surface water data to conservative water column criteria.
> **ME2**- Compare valid water column bioassay results to reference bioassay results.

P-AE2: Survival, growth, and reproduction of aquatic benthic invertebrates.

P-RQ2: Do COPEC concentrations in sediments pose unacceptable incremental ecological risk to aquatic benthic invertebrates?

> **ME1**- Compare site-specific sediment data to sediment benchmarks.
> **ME2**- Compare valid benthic invertebrate bioassay results to San Francisco Bay amphipod survival reference envelope tolerance limit.

P-AE3: Survival, growth, and reproduction of fish.

P-RQ3: Do COPEC concentrations in sediments and surface waters pose unacceptable incremental ecological risk to fish?

> **ME1**- Compare site-specific sediment and water data to conservative sediment screening values and water column criteria.
> **ME2**- Compare valid fish bioassay results to reference sample results.

P-AE4: Survival, growth, and reproduction of herbivorous birds[2 above (p. 7-8)].

P-RQ4: Do COPEC concentrations in sediments and associated biota pose unacceptable incremental ecological risk to herbivorous birds?

> **ME1**- Compare modeled dietary site and reference COPEC exposure to toxicity reference values (calculate hazard quotients).

P-AE5: Survival, growth, and reproduction of benthic-feeding birds.

P-RQ5: Do COPEC concentrations in sediments and associated biota pose unacceptable incremental ecological risk to benthic-feeding birds?

> **ME1**- Compare modeled dietary site and reference COPEC exposure to toxicity reference values (calculate hazard quotients).

P-AE6: Survival, growth, and reproduction of fish-eating birds[2 above (p. 7-8)].

P-RQ6: Do COPEC concentrations in sediments and associated biota pose unacceptable incremental ecological risk to fish-eating birds?

> **ME1**- Compare modeled dietary site and reference COPEC exposure to toxicity reference values (calculate hazard quotients).

P-AE7: Survival, growth, and reproduction of omnivorous mammals.

P-RQ7: Do COPEC concentrations in sediments and associated biota pose unacceptable incremental ecological risk to omnivorous mammals?

> **ME1**- Compare modeled dietary site and reference COPEC exposure to toxicity reference values (calculate hazard quotients).

7.2.4 Selection of Representative Receptors

Because it is impractical to assess the toxic effects of COPECs to all potentially exposed ecological receptors, a receptor was chosen to act as a "surrogate species" for each AE. These representative receptors are analogous to the "representative species" defined by DTSC (1996b) as follows:

- Species that represent a functional group of organisms at the site; and,

- Species that are chosen based primarily on their function in the ecosystem and secondarily on taxonomic relatedness and known or presumed similarities in physiology and life history.

For those AEs that are not supported by species-specific measurement endpoints (specifically, U-AE1, U-AE2, U-AE8, W-AE1, W-AE2, W-AE10, P-AE1, P-AE2, and P-AE3), selection of representative receptors was not necessary. Therefore, selection of representative receptors was focused on AEs selected for upper trophic-level groups. The selection of representative receptors is not based on an analysis of relative sensitivity based on exposure factors, but representative receptors have been selected based on practical considerations with the goal of selecting sensitive receptors that are believed to be protective of the receptor groups they represent. Specifically, the following attributes were considered to select representative receptors:

- The species is commonly observed at the site. Species observed relatively often at the site are likely to use the site to a greater degree than species not commonly observed, and therefore are likely to be more representative;

- The receptor has a feeding strategy likely to maximize exposure to site-associated contamination and has diet preferences that are representative of the guild in question;

- The receptor has a small relative body size. Smaller species have larger contaminant-absorbing surfaces per unit volume than larger species (Suter, 1993);

- The species is a special-status species. Where a general receptor class includes one or more special-status species, either a special-status species was selected as the representative receptor, or a representative receptor was selected that is believed to be protective of the special-status species;

- The receptor has a relatively small home range. Animals with smaller home ranges are likely to spend more time foraging on the site than animals with larger home ranges; and

- The receptor-specific life history data (exposure parameters) are readily available.

The rationale used for selection of representative receptors for each habitat present at IR Site 2 is provided in the following subsections.

7.2.4.1   Upland

U-AE3: Survival, growth, and reproduction of <u>omnivorous birds</u>
Selected Representative Receptor: <u>Western meadowlark (*Sturnella neglecta*)</u>

Approximately 30 species of omnivorous birds have been observed in the upland areas of the site, including one special-status species, the California horned lark. This species was only observed on one of ten survey dates and only one individual was observed, whereas the western meadowlark was observed on five of ten survey dates in 1997, and was observed in relatively high numbers. The western meadowlark was selected as the representative receptor because it is common on site, has a small body size, is primarily an insectivore (the California horned lark's primary food is seeds), and life history data are readily available.

U-AE4: Survival, growth, and reproduction of <u>burrowing birds</u>.[2 above (p. 7-8)]
Selected Representative Receptor: <u>Burrowing owl (*Athene cunicularia*)</u>

Although the burrowing owl was not observed during bird surveys conducted during 1997, it was reported that at least one burrowing owl has been recently observed at IR Site 2 (Feinstein, 2004). Because the burrowing owl is a California Species of Special Concern, and it is potentially exposed to chemicals in environmental media through unique exposure pathways among birds (e.g., inhalation of soil gas potentially containing VOCs), the burrowing owl is included as a representative receptor.

U-AE5: Survival, growth, and reproduction of <u>carnivorous birds</u>.[2 above (p. 7-8)]
Selected Representative Receptor: <u>Northern harrier (*Circus cyaneus*)</u>

Four special-status and ten non-special-status terrestrial carnivorous birds have been observed on site. The northern harrier was selected as the representative receptor because it has been observed breeding on site, it is a California species of special concern, it is a year-round San Francisco Bay resident, and exposure parameters are readily available.

U-AE6: Survival, growth, and reproduction of <u>herbivorous mammals</u>.
Selected Representative Receptor: <u>California vole (*Microtus californicus*)</u>

Although the black-tailed hare, the domestic rabbit, and the Botta Pocket Gopher were the only herbivorous mammals observed on site during the 1997 surveys, other herbivorous mammals that are smaller may use the site, and may not be conservatively protected by an exposure model for a larger animal. Therefore, the vole was selected as the representative receptor. No special-status terrestrial herbivorous mammals were identified as inhabiting or potentially inhabiting the site.

U-AE7: Survival, growth, and reproduction of <u>omnivorous mammals</u>.
Selected Representative Receptor: <u>Raccoon (*Procyon lotor*)</u>

The raccoon is the only omnivorous mammal that has been identified inhabiting wetland areas on site and is assumed to also forage in upland areas. No special-status omnivorous mammals have been identified inhabiting or potentially inhabiting the site. Therefore, the raccoon was selected as the representative receptor for this AE.

U-AE9: Survival, growth, and reproduction of <u>carnivorous mammals</u>.
Selected Representative Receptor: <u>Red fox (*Vulpes vulpes*)</u>

Although the red fox was not observed on site during the 1997 surveys, the appropriate habitat does exist on site, and it may be expected to be present at the site in the future. The red fox was selected as the representative receptor because it has exposure parameters that are readily available. No special-status terrestrial carnivorous mammals have been identified inhabiting or potentially inhabiting the site. The red fox currently is managed to protect the nearby least tern community.

7.2.4.2   Wetland

W-AE3: Survival, growth, and reproduction of <u>omnivorous birds</u>.[2 above (p. 7-8)]
Selected Representative Receptor: <u>Alameda song sparrow (*Melospiza melodia pusillua*)</u>

Three special-status species and 15 non-special-status non-probing omnivorous birds have been observed using the wetlands at the site. The Alameda song sparrow, one of the special-status species observed on site, was selected as the representative receptor because it was observed in the wetlands area of the site during both survey events, and natural history information is readily available.

W-AE4: Survival, growth, and reproduction of <u>benthic-feeding birds</u>
Selected Representative Receptor: <u>Least sandpiper (*Calidris minutilla*)</u>

A probing bird was selected as the representative receptor because probers have the highest potential for direct uptake of contaminants in sediment in addition to exposure through prey items. The least sandpiper has been commonly observed at the site (during seven of ten survey events in 1997), it has a relatively small body size, and exposure parameters are readily available.

W-AE5: Survival, growth, and reproduction of <u>burrowing birds</u>[2 above (p. 7-8)]
Selected Representative Receptor: <u>Burrowing owl (*Athene cunicularia*)</u>

The burrowing owl could be present in the wetland as well as in the upland area of the site. For reasons noted in the previous section, the burrowing owl was selected as a representative receptor, and risk from exposure to chemicals present in the wetland area also will be assessed for the burrowing owl.

W-AE6: Survival, growth, and reproduction of <u>carnivorous birds</u>.[2 above (p. 7-8)]
Selected Representative Receptor: <u>Northern harrier (*Circus cyaneus*)</u>

The northern harrier could be present in the wetland as well as in the upland area of the site. The northern harrier was selected as the representative receptor because it has been observed breeding on site, it is a California species of special concern, it is a year-round San Francisco Bay resident, and exposure parameters are readily available. Risk from exposure to chemicals present in the wetland area also will be assessed for the northern harrier.

W-AE7: Survival, growth, and reproduction of <u>herbivorous mammals</u>.
Selected Representative Receptor: <u>California vole (*Microtus californicus*)</u>

The California vole could be present in the wetland as well as in the upland area of the site. For reasons noted in the previous section, the California vole was selected as a representative receptor, and risk from exposure to chemicals present in the wetland area also will be assessed for the California vole.

W-AE8: Survival, growth, and reproduction of <u>omnivorous mammals</u>.
Selected Representative Receptor: <u>Raccoon (*Procyon lotor*)</u>

The raccoon could be present in the wetland as well as in the upland area of the site. For reasons noted in the previous section, the raccoon was selected as a representative receptor, and risk from exposure to chemicals present in the wetland area also will be assessed for the raccoon.

W-AE9: Survival, growth, and reproduction of <u>carnivorous mammals</u>.
Selected Representative Receptor: <u>Red fox (*Vulpes vulpes*)</u>

The red fox could be present in the wetland as well as in the upland area of the site. For reasons noted in the previous section, the red fox was selected as a representative receptor, and risk from exposure to chemicals present in the wetland area also will be assessed for the red fox.

7.2.4.3   Wetland Ponds

P-AE4: Survival, growth, and reproduction of <u>herbivorous birds</u>.[2 above (p. 7-8)]
Selected Representative Receptor: <u>Mallard (*Anas platyrhynchos*)</u>

The mallard was observed during all ten bird survey events in 1997, and like the Canada goose it prefers to eat plant material and algae when available. Due to its smaller body size, the mallard can be used as a conservative surrogate for the Canada goose, a federal threatened species. Additionally, exposure parameters are readily available for the mallard.

P-AE5: Survival, growth, and reproduction of <u>benthic-feeding birds</u>.
Selected Representative Receptor: <u>Least sandpiper (*Calidris minutilla*)</u>

The least sandpiper could be present in the pond as well as in the wetland area of the site. For reasons noted in the previous section, the least sandpiper was selected as a representative receptor, and risk from exposure to chemicals present in the pond area also will be assessed for the least sandpiper.

P-AE6: Survival, growth, and reproduction of <u>fish-eating birds</u>.[2 above (p. 7-8)]
Selected Representative Receptor: <u>Great blue heron (*Ardea herodias*)</u>

Although historical bird survey data indicate that a large Caspian tern breeding colony is present on site, evidence indicates habitat change in the wetlands has induced this colony to move off site, and the site is not believed to support Caspian tern. A least tern breeding colony is located on the former runway area to the west and northwest of the site. Although this protected least tern colony (which is managed by USFWS) is located near IR Site 2, the least terns are not shown or expected to feed at the site. The use of the offshore areas of Alameda Point by the least tern colony has been documented in a 10-year observational study (Collins and Feeney, 1995), and it was shown that least terns feed off-site 100% of the time. The 1994 and 1997 bird surveys (see Section 2.0 of this RI Report for additional details on bird survey results) did not indicate any use of the site by the least tern.

The great blue heron, another fish-eating bird, is expected to forage in the wetland ponds if/when food is available. The great blue heron was observed on site on five of the ten survey dates in 1997. Special-status species that may feed in the vicinity of the wetland include white pelican, brown pelican, and double-crested cormorant. Due to the limited size and shallow nature of the wetland ponds, pelicans are not expected to obtain a significant portion of their food in the wetlands at the site. The relatively similar foraging strategy and body size of the great blue heron and the double-crested cormorant support the assumption that an exposure-response conservatively developed for the great blue heron would be protective of the double-crested cormorant.

P-AE7: Survival, growth, and reproduction of <u>omnivorous mammals</u>.
Selected Representative Receptor: <u>Raccoon (*Procyon lotor*)</u>

Although no mammalian species were observed using the wetland ponds on site, the raccoon is opportunistic and may forage on invertebrates and amphibians that are associated with the wetland ponds. For reasons noted previously, the raccoon was selected as a representative receptor, and risk from exposure to chemicals present in the pond area also will be assessed for the raccoon.

7.2.5 Preliminary COPEC Selection

All chemicals detected at the site were selected as COPECs for the screening-level ERA. COPECs that do not have available risk-based screening criteria were retained as potential risk drivers. Some chemicals that are related also have been evaluated as sums, such as PCBs; PAHs; DDT, DDD, and DDE; and PCDD/PCDF. Sums of individual samples were calculated using the raw analytical data, and then statistics (e.g., FOD, maximum concentration and 95% UCL) were calculated for the summed chemicals. The methods used for summing PCBs; DDT, DDD, and DDE; and PCDD/PCDF are described in Section 5.1. The summation of PAHs is described in the following paragraph.

Individual PAHs were evaluated using screening benchmarks and toxicity reference values (TRVs), as available. Avian and mammalian TRVs are available for each of the 16 individual priority pollutant PAHs. PAHs also were evaluated as the sums of two subgroups based on the molecular weight of the individual PAHs. Sums of low-molecular-weight PAHs (LPAHs: acenaphthene, acenaphthylene, anthracene, fluorene, naphthalene and phenanthrene) and high-molecular-weight PAHs (HPAHs: benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(g,h,i)perylene, benzo(k)fluoranthene, chrysene, dibenz(a,h)anthracene, fluoranthene, indeno(1,2,3)pyrene, and pyrene) were evaluated. Only those PAHs that were reported above the MDL were included in the summation.

The U.S. EPA states in its Eco-SSL document for aluminum (U.S. EPA, 2005g) that aluminum must be in a soluble form to be toxic to biota, and the forms of aluminum that may be toxic are poorly understood. It is difficult to measure with precision or accuracy the concentration of soluble aluminum in soil extracts, and use of a total aluminum concentration in soil is inappropriate for assessing toxicity. As an alternative, U.S. EPA recommends the measurement of soil pH as "an indirect but reliable approach for assessing if soluble aluminum could be present. The use of a pH screening-level of [at least] 5.5 is considered environmentally protective" (U.S. EPA, 2005g). U.S. EPA identifies aluminum as a COPEC only for those soils with a soil pH less than 5.5, because the soluble and toxic forms of aluminum are only present in soil at pH values of less than 5.5. It is not expected that soil pH at IR Site 2 is less than 5.5, and there are no known sources of aluminum at the site other than the native concentrations in site soils. Therefore, aluminum was not selected as a COPEC.

7.2.6 Data Quality Assessment Summary

Analytical data generated through off-site laboratory analysis of the samples collected at IR Site 2 and the project reference areas were validated in accordance with the protocols described in the project SAP (i.e., Appendix A of the Work Plan [Battelle et al., 2005]). The data and validation results were reviewed and found to meet the DQOs established for the RI. The data review and evaluation process assessed all analytical data generated by the RI site investigations summarized in Section 4.0 of this report for completeness and usability in the ERA and generated statistical summaries of the data by constituent and sample media. Only data of sufficient quality were accepted for use in the ERA. The results of the data review and evaluation process are discussed previously in Section 5.0 and in Appendix G of this RI Report. The data now available representing the various environmental media at IR Site 2 are sufficient in both quantity and quality to perform the ERA.

7.3 Screening-Level Assessment

This section presents the screening-level assessment and includes a discussion of approaches used to assess exposure and ecological effects. Results are presented in the form of hazard quotients, and are summarized for each habitat and receptor group identified as part of the problem formulation described in Section 7.2. A characterization of risk is not included in the assessment because screening-level exceedances do not provide evidence of risk. Instead, the results of the assessment are used to focus the BERA on those compound-receptor pairs that fail the conservative screen. The results of the screening-level assessment presented in this section are thus discussed in greater detail and interpreted in Section 7.4.1 to focus the BERA.

7.3.1 Screening-Level Exposure Assessment

This section presents the approaches used to develop screening-level exposure estimates for the upland, wetland, and wetland ponds habitats. The general approach was to incorporate considerable conservatism into the development of exposure estimates in order to minimize the potential for falsely screening a COPEC from further evaluation based on the screening-level assessment. Two types of exposure estimates were included in this screening-level evaluation:

- Exposure to COPECs in abiotic media via direct contact (including exposure to COPECs via inhalation (applies to burrowing animals only) and radiation; and,

- Exposure to COPECs via uptake through the food web.

Significant conservatism is incorporated into each of these types of exposure estimates. For example, maximum detected site concentrations are used to represent COPECs concentrations in abiotic media in each habitat type, receptors are assumed to live and forage exclusively within the boundaries of the site, and conservative uptake or transfer factors are used to estimate concentrations of COPECs in site associated biological tissues. The ecological features of the upland, wetland, and wetland ponds exposure units have been described in previous sections of this RI Report. Exposure estimates developed for these three separate areas (upland, wetland, and wetland ponds) are calculated using data collected from the respective areas. Although data were collected during two seasons (i.e., wet season and dry season), this ERA does not provide an assessment that specifically addresses seasonality. Where seasonal data exist, they have been combined to ensure that screening-level exposure estimates based on maximum detected concentrations (for screening-level estimates) and baseline exposure estimates capture the full range of conditions experienced by ecological receptors over different seasons. In addition to upland, wetland, and wetland ponds boundaries, exposure estimates were developed for the following receptors using data collected from the indicated depth strata:

- **0 to 1 ft bgs**: Upland and wetland invertebrates; and upland and wetland birds and mammals

- **0 to 6 ft bgs**: Upland plants

- **0 to TD**, which corresponds to the top of the groundwater table: Wetland plants

- **0 to 0.3 ft bgs**: Benthic invertebrates; and birds and mammals that feed in wetland ponds

- **Top 1 ft of the surface water column**: Water column invertebrates and fish.

7.3.1.1 Calculation of Exposure Point Concentrations (EPCs) for Screening-Level Direct Contact Evaluation

Soil, sediment, and surface water data collected from IR Site 2 were used to develop screening-level EPCs for receptors exposed directly to chemicals in these media (i.e., plants, invertebrates, aquatic invertebrates, and benthic invertebrates). For the screening-level evaluation, maximum detected concentrations were used as EPCs for upland media, wetland media, and wetland pond media. These exposure estimates are presented in the following tables:

- Upland screening-level soil EPCs for direct contact endpoints are presented in Tables K-1 (0-1 ft bgs) and K-2 (0-6 ft bgs) in Appendix K-1.

- Wetland soil EPCs for direct contact endpoints are presented in Tables K-3 (0-1 ft bgs) and K-4 (0-6 ft bgs) in Appendix K-1.

- Exposure by direct contact was assessed separately for North Pond and South Pond in the screening-level ERA. Wetland pond sediment and surface water EPCs for direct contact endpoints are presented in Tables K-5 (North Pond sediment), K-6 (South Pond sediment), K-7 (North Pond surface water), and K-8 (South Pond surface water) in Appendix K-1.

The two burrowing receptors selected for evaluation (California vole and burrowing owl) may be exposed to VOCs in air while underground. Soil gas data collected from the landfill were used to develop screening-level EPCs for this exposure pathway. Although the ERA study design includes assessment of burrowing animals in both the upland and the wetland areas, soil gas data were only available for the upland portion of the landfill. Therefore, screening-level exposure estimates for burrowing receptors could only be developed for the upland area. To develop upland exposure estimates, the maximum concentration of any VOC detected at least once in quarterly soil gas samples collected from March 2003 to March 2005 were used in the screening-level assessment. These EPCs were converted from units of parts per billion by volume (ppbv) to micrograms per cubic meter ($\mu g/m^3$) and compared to TRVs for concentrations in air. Table K-9 in Appendix K-1 presents the maximum soil gas COPEC concentrations used for the screening-level evaluation of the subsurface inhalation pathway for burrowing receptors.

The maximum detected soil, sediment, and surface water concentrations for radionuclides were used as screening-level EPCs for aquatic and terrestrial receptors at the site. The screening-level ionizing radiation EPCs are summarized for the upland, wetland, and the wetland pond sediment and surface water habitats in Tables K-10, K-11, K-12, and K-13, respectively, in Appendix K-1.

7.3.1.2 Screening-Level Food Web Exposure Estimates

This section describes the model and associated inputs that were used to estimate ingested doses of site-related COPECs for birds and mammals using measured and estimated concentrations in prey items. Exposure models estimate the mass of chemical internalized daily by a receptor per kilogram of body weight (hereafter referred to as the "daily chemical dose"). Estimates of exposure are generally based on knowledge of the spatial and temporal distribution of both chemicals and receptors and on specific natural and life history characteristics that influence exposure to chemicals. Site-specific data on chemical concentrations in soil, sediment, and prey items are used in conjunction with available literature values for receptor-specific parameters to estimate the daily chemical dose. The exposure model for birds and mammals at the site assumes that exposure to chemicals is primarily through ingestion of contaminated soil, sediment, and prey. Therefore, other possible routes of exposure, such as surface water ingestion, dermal contact, and inhalation, were not evaluated using this model. An evaluation of the inhalation

pathway for burrowing animals is included in this ERA, but a different exposure model is used to conduct this assessment. These exposure routes are generally not significant relative to exposure through ingestion of soil, sediment, and prey.

Daily chemical doses were estimated for each COPEC and representative receptor where adequate data were available and these models were appropriate, using conservative assumptions for the screening-level ERA. These doses then were compared to low TRVs to estimate the potential adverse biological effects on the receptor. Based on this comparison, the risk to each representative receptor was characterized using a HQ.

The total exposure from ingestion of soil or sediment and food items by each receptor of concern was calculated as the sum of the dietary exposure estimates. The general equation used to calculate food chain exposure of receptors (in mg/kg body weight/day) is as follows:

$$Dose = \frac{\left[\left(C_{soil/sed} \times IR_{soil/sed}\right) + \left(C_{prey\,1} \times P_{prey\,1} \times IR_{food}\right) + \left(C_{prey\,2} \times P_{prey\,2} \times IR_{food}\right) + \left(C_{prey\,3} \times P_{prey\,3} \times IR_{food}\right)\right] \times SUF}{BW} \quad (7\text{-}1)$$

where:

Dose	=	daily dose resulting from ingestion of soil or sediment and each food type (mg COPEC/kg body weight/day)
$C_{soil/sed}$	=	COPEC-specific concentration in surface soil or sediments (mg COPEC/kg soil or sediment)
$C_{prey\,1,2,3}$	=	COPEC-specific concentration in each food type (mg COPEC/kg tissue)
$P_{prey\,1,2,3}$	=	proportion of each prey type in receptor diet (kg prey/kg food)
$IR_{soil/sed}$	=	estimate of receptor's daily incidental ingestion rate of soil or sediment (kg soil or sediment/day)
IR_{food}	=	estimate of daily ingestion rate of food (kg food/day)
SUF	=	site use factor (unitless)
BW	=	body weight (kg).

Every effort was made to select ecologically relevant and conservative ingestion rates, body weights, and dietary compositions. To ensure conservatism in the screening-level ERA, an SUF of 1 was assumed for each receptor, which means that it was assumed that each receptor obtains 100% of its prey from the site (the SUF for some avian receptors in the baseline assessment is adjusted based on foraging range, as shown in the following sections). Receptor-specific exposure parameters for each representative receptor, such as dietary proportions ($P_{prey\,1,2,3}$), incidental soil or sediment ingestion rates ($IR_{soil/sed}$), ingestion rates of food (IR_{food}), body weight (BW), and site use factors (SUF) are discussed in the following section.

The evaluation of risk to birds and mammals is based on the selected AEs and MEs identified in Section 7.2.3. Food web models are used to estimate the exposure of birds and mammals to chemicals in their diet, as described in the previous section. These models are conceptually simple and focus on the selected representative ecological receptors. Food web models are one method of integrating ecological and chemical information into the risk assessment process, especially for chemicals that tend to biocon-centrate or bioaccumulate (Pascoe et al., 1996). Receptors representing the various feeding guilds of birds and mammals were selected as described in Section 7.2.4.

Exposure was assessed within the context of the following linear food chains to evaluate potential ecological effects on primary, secondary, and tertiary consumer bird and mammal species:

Landfill Soil → Plants and Invertebrates → Western Meadowlark
Landfill and Wetland Soil → Invertebrates and Small Mammals → Burrowing Owl
Landfill and Wetland Soil → Small Mammals → Northern Harrier
Landfill and Wetland Soil → Plants → California Vole
Landfill and Wetland Soil → Plants, Invertebrates, and Small Mammals → Raccoon
Landfill and Wetland Soil → Plants, Invertebrates, Small Mammals and Birds → Red Fox
Wetland Soil → Plants and Invertebrates → Alameda Song Sparrow
Pond Sediment and Wetland Soil → Invertebrates → Least Sandpiper
Pond Sediment → Plants and Invertebrates → Mallard
Pond Sediment → Invertebrates and Fish → Great Blue Heron

Daily doses are estimated based on average natural history parameters and conservative transfer coefficients found in the literature. The natural history parameters were selected from the literature based on the most recent data and regional locations of the studies with the information. For instance, the body weight and foraging range for the burrowing owl was selected from a study conducted at the Oakland Airport, which is very close to Alameda Point. For upland species, the site exposure area is the landfill, which totals approximately 77 acres. For most wetland receptors, the site exposure area includes the entire wetland area (including the ponds), which totals approximately 33 acres.

Tables 7-1 through 7-17 present species-specific exposure parameters used to conduct the screening-level assessment.

7.3.1.2.1 Estimation of Tissue EPCs for Use in Food Web Model

When available, site-specific media measurements for soil, sediment, and upland and wetland plants were used to derive EPCs for food web models. Maximum detected concentrations (or one-half of the maximum detection limit for COPECs with 0% FOD) were used in the dose calculations for the screening-level ERA. Summary statistics and EPC estimates for soil, sediment, and plants are summarized in Tables K-1 through K-6 in Appendix K-1. Upland screening-level plant EPCs are presented in Tables K-14 (upland plants) and K-15 (wetland plants). Sufficient terrestrial invertebrate (upland and wetland), aquatic benthic invertebrates, fish, and small mammal tissue could not be collected in sufficient quantities to support chemical analysis (see Appendix B of this RI Report for discussion of modifications to the field study plan); therefore, measured data for site-associated COPECs are not available for these media.

When site-specific media measurements were not available to derive tissue EPCs for food web modeling, models were used to estimate uptake into prey. Depending on the availability of site-specific or Alameda-specific data, the following hierarchy of approaches was used to estimate prey tissue concentrations at the site:

1. Site-specific bioaccumulation assay results

2. Site-specific or Alameda-specific bioaccumulation factors (BAFs)

3. Non-linear regression or linear uptake models developed by the Oak Ridge National Laboratory (Sample et al., 1998b)

4. Non-linear regression or linear uptake models developed by U.S. EPA (2005a)

5. Use of other literature sources, surrogate values, and conservative assumptions.

The application of each of these approaches to specific tissue is summarized in Table 7-18. The method for modeling uptake into prey tissue varied by constituent and by media. Due to the extensive list of constituents analyzed in site samples, there are many compounds for which analytical data are available, but for which neither regression models nor uptake factors (UFs) have been derived. A description of each of these approaches is provided below, and the following tables present the EPCs resulting from the application of these approaches:

- Upland screening-level plant EPCs for ingestion endpoints are presented in Table K-14.

- Upland screening-level invertebrate EPCs for ingestion endpoints are presented in Table K-16.

- Upland screening-level small mammal EPCs for ingestion endpoints are presented in Table K-17.

- Wetland screening-level plant EPCs for ingestion endpoints are presented in Table K-15.

- Wetland screening-level invertebrate EPCs for ingestion endpoints are presented in Table K-18.

- Wetland screening-level small mammal EPCs for ingestion endpoints are presented in Table K-19.

- Wetland ponds screening-level plant EPCs for ingestion endpoints are presented in Table K-15.

- Wetlands pond screening-level invertebrate EPCs for ingestion endpoints are presented in Tables K-20 and K-21.

- Wetlands pond screening-level fish EPCs for ingestion endpoints are presented in Table K-22.

Site-Specific Bioaccumulation Bioassays

Site-specific laboratory bioaccumulation bioassays were used to develop exposure estimates for wetland pond benthic invertebrates in order to provide the necessary inputs to food web models for benthic-feeding birds, omnivorous mammals, and fish-eating birds. The clam *M. nasuta* and the polychaete worm *N. caecoides* were exposed to sediments for 28 days to determine whether the body burdens of contaminants of potential environmental concern pose an unacceptable risk to these upper trophic-level receptors. This 28-day test is a standard bioaccumulation bioassay protocol. The two organisms selected are conservative surrogates to assess contaminant uptake by benthic invertebrates. As they live and feed in the sediments, they reach equilibrium with contaminants relatively quickly, and they have relatively high lipid content; thus, bioaccumulative compounds accumulate at relatively high concentrations in their tissues. Tissue residues from site samples are used to represent prey item-associated contaminant exposure using a food web model. This section summarizes the results of these bioaccumulation bioassays. Because the primary use of bioaccumulation tests was to provide adequate tissue mass for analytical chemistry to evaluate tissue burdens of COPECs, survival was included in the stated decision rules for this study only to confirm the validity of the testing protocol. The study did yield sufficient tissue volume to conduct some laboratory chemical analysis of tissues (see Appendix B).

Survival of *M. nasuta* exposed to IR Site 2 sediment samples generally ranged from 76.7% to 96.7%. *M. nasuta* survival was 0% for sediment sample SED16 from the South Pond (see Figure 3-5 and Table 3-4 in Appendix C). As discussed in Section 2 of Appendix C, SED16 had the highest initial porewater salinity (98‰) of any sediment sample evaluated (Table 2-3 in Appendix C). It is possible that the high level of porewater salinity affected *M. nasuta* survival for this sample. Survival of *M. nasuta* in the native control sediment exposure was 76.7% and 66.7% in replicates 1 and 2, respectively, and survival of *M. nasuta* exposed to CCSP reference sediment and soil samples ranged from 60 to 93.3%. The control survival displayed is below the test-specific criteria of ≥80% survival (Table 3-4 of Appendix C). The reason for low control survival is not known and is problematic given the high survivals observed in other sediment samples. It is possible the soil matrix was inappropriate for this species and influenced test organism survival. It is not possible to determine whether low *M. nasuta* control survival influenced contaminant uptake during the 28-day bioaccumulation exposure, but organisms surviving the 28-day exposure appeared to be healthy and unaffected.

Survival of *N. caecoides* generally ranged from 53.3 to 100% except for sediment samples SED15, SED16, and SED17 (all South Pond samples), for which survival was 30% or less (see Figure 3-6 and Table 3-4 in Appendix C). Unlike the clam exposures, *N. caecoides* survival was quite high for sediments and soil. Survival of *N. caecoides* in the control exposure was acceptable. It is likely the high porewater salinity contributed to the low survivals observed in some stations for both species, and seasonal fluctuations in *M. nasuta* sensitivity contributed to the observed control response. Survival was evaluated for the bioaccumulation study only to assess the impact of sediment and soil exposures on the test organisms. As mentioned, the primary purpose of the bioaccumulation exposures was to provide adequate tissue mass to evaluate tissue burdens of contaminants. The study did generally yield sufficient tissue volume to conduct laboratory chemical analysis of tissues, although some prioritization of analysis was required as tissue volume was not sufficient for depurated and non-depurated treatments of both species to support all planned chemical analyses. Refer to Table B-2 in Appendix B for a summary the analyses that were performed on tissue samples originating from the bioaccumulation testing of site-specific media.

Tissue analytical results from depurated bioaccumulation bioassay test organisms were used to support the development of tissue EPCs for dose modeling. The use of depurated results is considered appropriate as incidental sediment ingestion is accounted for in the dose models, and sediment ingestion rates generally are estimated based on studies which measure the acid-insoluble ash content of wildlife scats or digestive tract contents. Using these approaches, sediment incidentally ingested during feeding and/or other behaviors (e.g., preening) and sediment associated with prey (i.e., both adhering on the outside as well as within the alimentary canal) are accounted for in the dose model.

Depurated results were used for both species that originated from the bioaccumulation tests. The test species displaying the highest uptake on a compound-specific basis was used to develop tissue EPCs for wetland pond benthic invertebrates. EPC calculation inputs and resulting EPCs are provided in Table K-20 and K-21. Summary statistics of the tissue data are provided in Tables K-23 and K-24.

Site-Specific or Alameda-Specific BAFs

To estimate tissue concentrations in prey, site-specific or Alameda-specific data were used in a "ratio estimate" approach to develop chemical-specific UFs. A ratio estimate (Cochran, 1963) is a statistic used to estimate the rate or ratio between two variables, both of which vary from sample to sample.

The ratio estimate is the ratio of the averages of two variables, in this case paired soil or sediment and tissue concentration results:

$$BAF_{soil/sediment\ to\ prey} = (average\ prey\ concentration/average\ soil\ or\ sediment\ concentration) \quad (7\text{-}2)$$

The ratio estimate has the advantage of being less variable and less biased than the average of the individual sample location ratios. It is preferred over regression-based analyses that have slopes dominated by a single influential point, which is a tendency that is even more pronounced when the regression is modeled through the origin to obtain a simple rate based on slope alone (Cochran, 1963). The UF attained using the ratio estimate is multiplied by the COPEC soil or sediment concentration to estimate a tissue concentration (wetland invertebrate and fish) as indicated by Equation 7-3.

$$C_{prey} = C_{soil} \times UF_{soil\ to\ prey} \quad (7\text{-}3)$$

where: C_{prey} = modeled COPEC concentration in prey tissue
C_{soil} = COPEC EPC for site soil
$UF_{soil\ to\ prey}$ = the uptake factor between soil and prey.

Wetland Invertebrates

As discussed in Appendix B, sufficient wetland invertebrate tissue could not collected from the site to support chemical analysis. Specifically, no site-specific wetland invertebrate tissue data were available to support the development of food web-based EPCs. To derive COPEC EPCs for wetland invertebrates, the ratio estimate method was applied to the tissue residue results of the benthic invertebrate bioassays to estimate compound- and site-specific UFs. For each COPEC detected in *M. nasuta* and/or *N. caecoides* tissue from the site-specific laboratory bioaccumulation bioassays, the ratio of the higher average *M. nasuta* tissue concentration and average *N. caecoides* concentration to the average sediment concentration was calculated. These calculated UFs were used to estimate wetland terrestrial invertebrate EPCs in the same manner as the ORNL UFs (i.e., they were multiplied by the wetland soil COPEC concentration).

Wetland terrestrial invertebrate UFs calculated with the ratio estimate are available for all constituents that were detected in *M. nasuta* and *N. caecoides* tissue. These constituent groups include metals, pesticides, PCBs, PAHs (except naphthalene, which was not measured), and other SVOCs. Constituent groups not analyzed in benthic invertebrate tissues include VOCs, PCDD/PCDF, explosives, and organotins. In addition, hexavalent chromium was not analyzed in benthic invertebrate tissue samples. The UF for invertebrates for naphthalene from U.S. EPA Eco-SSL documents (U.S. EPA, 2005a) was used because a UF could not be calculated using the ratio estimate method. Because VOCs are not considered to be bioaccumulative, a UF of 1.0 is considered conservative (U.S. EPA, 1995) and was used. The uptake regression models developed for TCDD by Sample et al. (1998a) were used to convert soil PCDD/PCDF TEQs to wetland invertebrate TEQs. No UFs or models were found and no surrogate values were selected for explosives or organotins for wetland invertebrates. Wetland terrestrial invertebrate UFs are summarized in Table K-18.

Wetland Pond Fish

As discussed in Appendix B, attempts to collect fish from the wetland ponds did not yield sufficient tissue to support chemical analysis. Specifically, no site-specific fish tissue data were available to support the development of food web-based EPCs. The ratio estimate method was previously used to determine site-specific sediment to fish UFs using forage fish data collected at Seaplane Lagoon, which is located southwest of IR Site 2 at Alameda Point. Several composite samples of forage fish (species commonly consumed by piscivorous birds such as the great blue heron) were collected from

areas of Seaplane Lagoon in 2001. The result for each composite tissue sample was considered an individual result for purposes of averaging. Sediment samples collected from Seaplane Lagoon during four sampling events within the areas where trawling was conducted to collect fish were used to derive average sediment concentrations. Note that all sediment samples collected from the lagoon were used in the sediment average, because the forage fish use areas smaller than the lagoon, and sediment samples were not collected concurrently with the fish collection[3].

The ratio of the average fish tissue concentration and the average sediment concentration was calculated for metals, PAHs, total PCBs, and pesticides analyzed in the forage fish. Sediment to fish UFs were calculated by this method for 12 metals, 17 PAHs, LPAHs and HPAHs, total PCBs, and 22 pesticides. UFs for all the metals, PAHs, and total PCBs are based on actual detected concentrations in fish tissue. Only six of the pesticides (including 4,4'-DDT, 4,4'-DDE and 4,4'-DDD) were detected in fish tissue. Six pesticides detected in wetland ponds at IR Site 2 were not detected in forage fish tissue in Seaplane Lagoon (2,4'-DDD, 2,4'-DDE, heptachlor epoxide, endosulfan II, endosulfan sulfate, and methoxychlor), resulting in UFs being calculated using the average of one-half the MDLs reported in the fish samples for these pesticides. Of these, heptachlor epoxide and methoxychlor were not detected in any sediment samples, and the endosulfans were only detected in one sediment sample. These UFs were multiplied by sediment concentrations from the North and South Ponds to determine EPCs in fish present in the ponds. No other methods were used to estimate constituent concentrations in fish from the wetland ponds. The sediment and fish data used to develop sediment-to-fish UFs are summarized in Tables K-25 and K-26. Calculated fish UFs and resulting fish EPCs are presented in Table K-22.

ORNL Regression Uptake Models and Uptake Factors

ORNL developed models for accumulation of constituents from soil into biota (Sample et al., 1998a and 1998b), including soil invertebrates (earthworms) and small mammals, by compiling numerous studies from several countries. Using these databases, ORNL developed bioaccumulation models (linear UFs and non-linear regression models). Their approach, applied separately to the earthworm data and small mammal data, was to include the data from most of the studies to develop the model for each constituent, and use independent data (not included in the model development) to evaluate the accuracy of concentrations estimated by the models. This validation step tested the reliability of the models. After the validation step, the data used for validation were incorporated into the database for calculation of final UFs and regression models. ORNL presents models or UFs for both general and conservative estimates of earthworm and mammal tissue concentrations (Sample et al., 1998a and 1998b). Those models or UFs developed for conservative estimates are appropriate for use in a screening-level evaluation and were used in both the screening-level and baseline risk evaluations.

Soil Invertebrates

For earthworms, which are considered in this ERA to be representative of terrestrial soil invertebrates, ORNL found that bioaccumulation is non-linear, decreasing as soil concentration increases. Thus, the use of log-linear regression models to estimate earthworm bioaccumulation is recommended by ORNL for most chemicals, with the exception of four metals for which UFs are recommended. These models and UFs are presented in Sample et al. (1998a) with details on the studies from which data were compiled and the statistical analyses conducted to validate the models.

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[3] The UFs presented in this document reflect the initial approach developed for Sea Plane Lagoon, and the Navy is currently considering new data collected in Sea Plane Lagoon to support investigations focused on the offshore sediments. The results of this effort are not yet available. If in the future the team assessing Sea Plane Lagoon and the offshore sediments at Alameda Point present a re-calculation of fish uptake factors based on more recent data collected in Sea Plane Lagoon, it may become necessary to revisit the inputs used in this RI.

Small Mammals

The mammal database included data from studies for mammals with various feeding habits including herbivorous, insectivorous and omnivorous trophic groups. Data from all studies were used to develop a regression model or UF for mammals of any trophic group, if possible. In some cases, better fits could be made using data from each trophic group separately, resulting in three models for a chemical, one for each trophic group. These models and UFs are presented in Sample et al. (1998b) with details on the studies from which data were compiled and the statistical analyses conducted to validate the models. For chemicals where separate models were presented for the three trophic groups, the model for omnivorous small mammals was used in this ERA because model concentrations in small mammal prey items from any or all of the trophic groups were necessary. Although some prey species at the site may be exclusively herbivorous, use of the omnivorous model includes invertebrates in the diet of prey species and is likely conservative, because invertebrates tend to bioaccumulate constituents to higher concentrations than plants do. The general regression model equation that was used is presented below (Sample et al., 1998a, 1998b):

$$\ln(C_{prey}) = a + b[\ln(C_{soil})] \tag{7-4}$$

where: C_{prey} = modeled COPEC concentration in prey tissue
C_{soil} = COPEC EPC for site soil
a = intercept of regression model
b = slope of regression

To calculate the tissue EPC for use in the food web model, the UF is multiplied by the soil concentration to estimate a tissue concentration (see Equation 7-3).

U.S. EPA Eco-SSL Regression Uptake Models and Uptake Factors

For most inorganic chemicals, the U.S. EPA Eco-SSL document (U.S. EPA, 2005a) used the recommended regression uptake model or uptake factor for invertebrates and mammals from ORNL (Sample et al., 1998a, 1998b). Models or uptake factors were developed by U.S. EPA (2005a) for three additional inorganic chemicals, antimony, barium and beryllium, and these were used in the screening-level and baseline risk evaluations. Table K-16 summarize UFs and regression models used to calculate upland terrestrial invertebrate EPCs. The small mammal UFs used to develop EPCs for both upland and wetland small mammals are summarized (with the terrestrial invertebrate UFs) in Tables K-17 and K-19.

Use of Surrogate Data and Conservative Assumptions
for Uptake Models

Where tissue EPC estimates could not be developed using one of the previously discussed approaches, other sources were used to develop uptake factors including data from scientific literature, and the use of conservative assumptions including surrogate compound data. The following list summarizes the other sources of information that were used to support the development of exposure estimates:

- UFs were unavailable from Sample (1998a) or U.S. EPA (2005a) for VOCs and invertebrates. Because VOCs are not considered to be bioaccumulative, a UF of 1.0 is considered conservative (U.S. EPA, 1995) and was used.

- Due to a lack of data regarding bioaccumulation of VOCs into biota, models or uptake factors have not been developed. However, in U.S. EPA (1995), it is noted that VOCs are not considered to be bioaccumulative, and that a UF of 1 is a conservative assumption for VOC uptake into biota. Based on this assumption, a UF of 1.0 was used to estimate VOC concentrations in plant, invertebrate, and small mammal tissues. As noted above, U.S. EPA (2005a) used the same conservative assumption for uptake of antimony into invertebrate tissue.

- No uptake models or factors were available in the literature for molybdenum, so the molybdenum UF calculated using the ratio estimate with benthic invertebrate and sediment molybdenum concentrations was used. The same UF assumption used to estimate VOC uptake in terrestrial invertebrates was used to estimate VOCs uptake in small mammals (i.e., a UF of 1.0 was used [U.S. EPA, 1995]).

- No uptake models or factors were found for uptake of PAHs and explosives into small mammals. Because mammals readily metabolize PAHs and explosives, UFs of 0 were assumed for these constituents (U.S. EPA, 2005a).

- No uptake models or factors were found for molybdenum uptake into terrestrial invertebrates or small mammals. For terrestrial invertebrates, the molybdenum UF was calculated using the ratio estimate with benthic invertebrate and sediment concentrations. For mammals, the highest UF from the other metals (the UF for thallium) was applied.

- The UF used for total PCBs for small mammals is presented in the Baseline Ecological Risk Assessment prepared for the Allied Paper Superfund site on the Kalamazoo River (Michigan Department of Environmental Quality, 2003).

- U.S. EPA's Draft Exposure and Human Health Reassessment of 2,3,7,8-TCDD and Related Compounds (U.S. EPA, 2003) was reviewed to select uptake models or factors for transfer of dioxins from soil to plants. It is recognized that the air-to-plant transfer of PCDD/PCDF is critical to the terrestrial animal food chain, and that little soil-to-plant transfer occurs because this class of compounds sorbs tightly to soil (U.S. EPA, 2003). Trapp and Matthies (1995) (as cited in Part I, volume 3 of U.S. EPA, 2003) present a comprehensive air/soil-to-plant modeling system, and in their application of this approach to 2,3,7,8-TCDD, they neglect the soil-to-aboveground portion of their model. As noted elsewhere in volume 3 of U.S. EPA's Draft Dioxin Reassessment (U.S. EPA, 2003), PCDD/PCDF contamination of fruits and vegetables is thought to occur primarily via particle deposition or vapor adsorption onto outer layers with little penetration to inner portions. Plant uptake from the soil via the roots and translocation to aboveground plant parts is generally considered negligible, with the exception of zucchini and pumpkins (Hülster and Marschner, 1993 as cited in U.S. EPA, 2003). In these plant species, root uptake appears to occur and leads to a uniform concentration within the fruit. Neither these nor similar species occur at IR Site 2. Because of the negligible transfer of dioxins in soil to aboveground plant parts, and given that any air-to-plant transfer occurring results from ambient atmospheric dioxins, an uptake factor of 0 was assumed for transfer of PCDD/PCDF from soil to plant parts generally consumed by ecological receptors at the site.

- For summed constituents in all tissue types where measured tissue data from the site were lacking, uptake factors or regression models were selected from the individual constituents as surrogates. For LPAHs and HPAHs, the uptake factors for naphthalene

and benzo(a)pyrene, respectively, were used. Detected PCDD/PCDF congeners were converted to 2,3,7,8-TCDD TEQs for birds and mammals using applicable TEFs. The uptake regression models developed for TCDD by Sample et al. (1998a and 1998b) were used to convert soil TEQs to invertebrate and mammal TEQs. Available uptake models for DDT, DDD, and DDE varied depending on the biota being modeled. The most conservative uptake factor of the 6 DDT isomers was used to estimate tissue EPCs for total DDT.

7.3.1.2.2 Summary of Screening-Level Dose Estimates

Exposure to upper trophic level receptors was estimated using food web models to calculate receptor-specific daily doses. Estimated screening-level dose estimates are summarized in Table 7-19 for upland receptors, Table 7-20 for wetland receptors, and 7-21 for wetland ponds receptors. Detailed dose calculation tables for individual receptors can be found in Appendix K-2.

7.3.2 Screening-Level Effects Assessment

In the screening-level assessment, maximum detected COPEC concentrations were compared to media-specific toxicity benchmarks for direct contact evaluations, and were compared to doses (expressed in mg/kg/day) for food web evaluations. The effects assessment identifies toxicological effects data (e.g., soil and sediment effect values, water quality criteria, and TRVs) that are used as benchmarks to compare to the site exposure concentrations presented in Section 7.3.1. Benchmarks and TRVs were selected to be conservative estimators of potential toxic effects. In other words, benchmarks were selected to minimize the possibility of reaching a finding of *de minimus* risk when risk actually exists. Benchmarks used in the screening-level assessment are based on no observable adverse effect levels (NOAELs), or other levels associated with low level sublethal effects such as a concentration associated with a 10% reduction in a non-lethal endpoint (EC10). In any case, the benchmarks used in the screening-level assessment constitute conservative thresholds of effect. When a conservative estimate of exposure does not exceed a conservative or screening-level benchmark it can be concluded that deleterious effects to biota are not expected. The direct contact screening-level benchmarks and food web TRVs are discussed and presented in the following subsections.

7.3.2.1 Direct Contact Benchmarks

The potential effects associated with direct contact to impacted soil, sediment, and surface water are evaluated via direct contact toxicity benchmarks. Receptors at IR Site 2 that are potentially exposed to COPECs via direct contact pathways include plants, terrestrial invertebrates, sediment-associated biota, and aquatic biota. The screening-level benchmarks, which represent conservative (i.e., protective) concentrations below which it is extremely unlikely that adverse ecological effects will occur, are presented as the "low" benchmarks in Table 7-22. The high benchmarks presented in tables associated with this section were used in the BERA and are discussed in greater detail in Section 7.4.3.2. The same plant and invertebrate benchmarks were used to evaluate upland and wetland soils, which are summarized in Table 7-22.

7.3.2.1.1 Upland

Receptors potentially exposed to soil-associated COPECs via direct contact in upland habitat at the site include upland plants and terrestrial invertebrates.

Plants

The following sources were used in the following order of priority to identify screening-level benchmarks for plants:

1. U.S. EPA. 2005. Ecological Soil Screening Levels (Eco-SSLs). Available at http://www.epa.gov/ecotox/ecossl/. Accessed on July 25, 2005.

2. Efroymson, R.A., M.E. Will, G.W. Suter, and A.C. Wooten. 1997a. Toxicological Benchmarks for Screening Contaminants of Potential Concern for Effects on Terrestrial Plants. Oak Ridge National Laboratory, Oak Ridge, Tennessee.

3. CCME, 1999. "Canadian soil quality guidelines for the protection of environmental and human health: NAPHTHALENE/BENZO(A)PYRENE." Canadian Environmental Quality Guidelines. Winnipeg; Canadian Council of Ministers of the Environment.

U.S. EPA Eco-SSLs

The Eco-SSLs are screening values representing concentrations of contaminants in soil that are protective of ecological receptors that commonly come into contact with soil or ingest biota that live in or on soil (U.S. EPA, 2003). They were derived by a multi-stakeholder workgroup consisting of federal, state, consulting, industry, and academic participants led by the U.S. EPA's Office of Emergency and Remedial Response. The general approach for deriving these values included literature searches for all available plant chronic toxicity test data, prioritizing biomass production and physiological endpoints for plants. Toxicity parameters considered acceptable for deriving Eco-SSLs were the EC20 (20% Effect Concentration), the MATC (Maximum Allowable Threshold Concentration), and the EC10 (10% Effect Concentration). LC50s (Lethal Concentration – 50% effect) and EC50s (sublethal Effect Concentration – 50% effect) were not used because they were not considered sufficiently conservative for screening-level purposes.

Eco-SSLs were calculated using data from studies considered acceptable by calculating the geometric mean of all toxicity values for which sufficient data exist (> three data points). The resultant Eco-SSLs are considered applicable screening benchmarks for plants at IR Site 2 because they were developed with chronic data and were specifically developed to be applied at the screening stage of an ecological risk assessment (U.S. EPA, 2003). As the guidance states, the Eco-SSLs are values used to identify constituents that require further evaluation in the site-specific baseline ecological risk assessment that is completed according to specific guidance (U.S. EPA, 1997, 1998, and 1999).

ORNL, 1997a

Phytotoxicity benchmarks recommended by ORNL and developed by Efroymson (1997b) were used as screening-level benchmarks in cases where an Eco-SSL was not available. Benchmarks recommended by ORNL were developed by rank-ordering lowest observed effect concentration (LOEC) values and choosing a number that approximated the 10th percentile. Similar to the Eco-SSLs, these benchmarks were developed to provide a conservative means to conduct a screening-level evaluation of risk to plants.

CCME, 1999

The low benchmark for LPAHs is based on the lowest EC25 from direct soil contact studies to plants reported in the Canadian Council of Ministries of the Environment (CCME) naphthalene soil quality guidelines studies. CCME's literature search reported three separate EC25s for plant exposures to naphthalene via direct contact in soil. The lowest value was used as the low benchmark for the

screening-level evaluation of LPAHs at IR Site 2, with naphthalene serving as the surrogate for all LPAHs.

The low benchmark for HPAHs is based on the lowest no observed effect concentration (NOEC) (for rye, wheat, and corn) reported in CCME's benzo(a)pyrene soil quality guideline studies. Benzo(a)pyrene was used as the surrogate for HPAHs, because this was the only HPAH for which values were available.

Terrestrial Invertebrates

Screening-level benchmarks for soil invertebrates were used in the following order of priority from these references:

1. U.S. EPA. 2005. Ecological Soil Screening Levels (Eco-SSLs). Available at http://www.epa.gov/ecotox/ecossl/. Accessed on July 25, 2005.

2. Efroymson, R.A., M.E. Will, and G.W. Suter. 1997b. Toxicological Benchmarks for Screening Contaminants of Potential Concern for Effects on Soil and Litter Invertebrates and Heterotrophic Processes. Oak Ridge National Laboratory, Oak Ridge, Tennessee.

3. Sverdrup, L.E., T. Nelson, and A.E. Kelly. 2002. Soil ecotoxicology of polycyclic aromatic hydrocarbons in relation to soil sorption, lipophilicity and water solubility. Environmental Toxicology and Chemistry.

U.S. EPA Eco-SSLs

As described above, the receptor-specific Eco-SSLs derived by the U.S. EPA and their consortium of stakeholders are considered appropriate as screening-level benchmarks. The values were derived based on literature searches for all available soil invertebrate chronic toxicity test data, prioritizing reproduction, population, and growth endpoints. Toxicity parameters considered acceptable for deriving Eco-SSLs were the EC20, the MATC, and the EC10. LC50s and EC50s were not used because they were not considered sufficiently protective of ecological resources.

Eco-SSLs were calculated using data from studies considered acceptable by calculating the geometric mean of all toxicity values for which sufficient data exist (> three data points). The resultant Eco-SSLs are considered applicable screening benchmarks for soil invertebrates at IR Site 2 because they were developed with chronic data and were specifically developed to be applied at the screening stage of an ecological risk assessment (U.S. EPA, 2003). U.S. EPA Eco-SSLs were used as screening-level invertebrate benchmarks in all cases where they were available.

ORNL, 1997a

Terrestrial invertebrate benchmarks for earthworms recommended by ORNL and developed by Efroymson (1997b) were used as screening-level benchmarks in cases where an Eco-SSL was not available. As with the plant values, benchmarks recommended by ORNL were developed by rank ordering LOEC values and choosing a number that approximated the 10th percentile.

Sverdrup et al., 2002

Terrestrial invertebrate PAH benchmarks were based on studies conducted by Sverdrup et al. (2002). Sverdrup developed toxicity data for the effects of 16 PAHs on the survival and reproduction of the soil-dwelling springtail *Folsomia filmetaria* via soil exposures for 21 days. The low benchmark for the screening evaluation is based on a reproduction EC10 for the springtails. It is not necessary to

assign surrogate invertebrate benchmarks because screening criteria were available for most PAHs. These PAH benchmarks are considered conservative screening-level effects thresholds and have been used in previous ERAs with PAH-impacted soils.

Effects Assessment for Radionuclides

The screening-level radiation risk assessment was conducted using a technical standard developed by the Department of Energy (DOE-STD-1153-2002), "A Graded Approach for Evaluating Radiation Doses to Aquatic and Terrestrial Biota" (DOE, 2002). This technical standard provides a graded approach including screening methods and methods for detailed analyses, and related guidance to evaluate whether radiation doses due to anthropogenic sources exceed specified limits of exposure to populations of aquatic animals, terrestrial plants, and terrestrial animals. Specifically, the technical standard provides dose evaluation methods that can be used to evaluate whether the dose limits for protection of biota developed or discussed by the NCRP (1991) and IAEA (1992) are exceeded. The graded approach uses the following biota dose limits:

- The absorbed dose to aquatic animals (fish, for purposes of this ERA) should not exceed 1 rad/d (10 mGy/d) from exposure to radiation or radioactive material releases into the aquatic environment (radioisotope concentrations in sediment and surface water).

- The absorbed dose to terrestrial plants (in the upland and wetland habitats) should not exceed 1 rad/d (10 mGy/d) from exposure to radiation or radioactive material releases into the terrestrial environment (radioisotope concentrations in soil).

- The absorbed dose to terrestrial animals (in the upland and wetland habitats) should not exceed 0.1 rad/d (1 mGy/d) from exposure to radiation or radioactive material releases into the terrestrial environment (radioisotope concentrations in soil).

For these receptor groups, biota concentration guidelines (BCGs) were developed for water, sediment, and soil for several radionuclides by back-calculating from the dose limits listed above. Decay products of each radionuclide with half-lives of less than 90 days are included in the BCG. Each radionuclide-specific BCG represents the limiting radionuclide concentration in the particular environmental medium that would not result in the recommended dose standards for biota being exceeded. The DOE has developed software (RESRAD-BIOTA version 1.0) for use in ecological assessment, with capabilities ranging from a screening-level assessment (at a habitat level, screening against dose limits for the most sensitive receptor class) to receptor-specific dose estimation. It was designed to be a tool for implementing the graded approach developed by DOE (2002).

The screening-level assessment for radionuclides is based on habitat type (i.e., terrestrial or aquatic). RESRAD-BIOTA calculates a ratio of the maximum measured radionuclide concentrations in the applicable environmental exposure media (i.e., water, sediment, and/or soil) and the associated BCGs. The sum of the ratios is calculated, because the DOE requirements and recommendations are based on the total weighted absorbed radiation dose rate from all radionuclides and pathways. If the sum exceeds one, additional assessment is needed. For conservatism, the ratios are calculated for the most sensitive receptor type (animal or plant) within a habitat type and then summed. Based on this method of summing, the following assessment endpoints were evaluated:

- Survival, growth, and reproduction of terrestrial receptors using the upland habitat;
- Survival, growth, and reproduction of terrestrial receptors using the wetland habitat; and,
- Survival, growth, and reproduction of fish inhabiting the wetland ponds and of terrestrial receptors that forage in the wetland ponds.

Six radionuclides (Pb-210, Ra-226, Ra-228, U-234, U-235, and U-238), gross alpha, and gross beta were tested in soil, sediment, and water samples. All were detected in upland soil, and all but Pb-210 were detected in wetland soil. All but Ra-228 were detected in sediment. Gross alpha was not detected in total or dissolved water samples, and U-235 was not detected in dissolved water samples. Radionuclide results for total water samples rather than dissolved water samples were used in the evaluation.

Because gross alpha and gross beta are measurements of types of radiation and RESRAD-BIOTA calculates ratios for specific radionuclides (for all radiation types emitted), gross alpha and gross beta are not directly used in the calculation. Of the six radionuclides tested in site samples, all but Pb-210 are included in RESRAD-BIOTA and were evaluated in the screening-level ERA.

Inhalation TRVs for Burrowing Animals

Low and high TRVs were developed to evaluate the potential for effects on burrowing animals from inhalation exposure to VOCs in soil gas. Limited inhalation toxicity data are available for avian species, and therefore, TRVs were not developed for burrowing birds, which are represented by the burrowing owl at IR Site 2. TRVs for the assessment of the inhalation pathway for burrowing mammals, which are represented by the California vole at IR Site 2, were selected from mammalian inhalation studies available from the toxicity data used to develop human health toxicity values as presented in the U.S. EPA's IRIS database, the ATSDR database, and Cal-EPA's toxicity database compiled by OEHHA.

General preferences for studies selected for development of small mammal inhalation TRVs included:

1) The study used small mammals;

2) The study was chronic or subchronic in exposure duration;

3) The toxicity endpoint was reproduction, growth or survival; and

4) Both a NOAEL and LOAEL were available from the study.

Consistent with other screening-level effects evaluations, NOAELs were generally preferred for developing low TRVs. LOAELs were preferred for developing high TRVs for the baseline risk evaluation.

Each selected NOAEL or LOAEL was converted from units of ppbv to units of mg/m^3 using Equation 7-5:

$$C_{sg}\,(\text{mg/m}^3) = \frac{C_{sg}\,(ppmv) \times \text{MW(g/mol)}}{\text{MGV(L/mol)}} \qquad (7\text{-}5)$$

where C_{sg} = Concentration of soil gas in units of milligram per cubic meter (mg/m^3)
C_{sg} = Concentration of soil gas in units of parts per million by volume (ppmv)
MW = Molecular weight in units of grams per mole (g/mol)
MGV = Molar gas volume in units of liters per mole (approximately 24.4 L for air at 25°C) (L/mol).

The C_{sg} then was converted to a low TRV using conversion factors to adjust for differences in exposure duration in the study and conservatively assumed exposure times for burrowing mammals. Specifically, two conversion factors were applied to adjust the exposure duration of the study (e.g., 5

days per week for 6 hours per day) to 7 days per week for 24 hours. A third conversion factor also was applied to C_{sg} to adjust the benchmark to account for the anticipated amount of time that the receptor spends in the burrow. Burrowing mammals were conservatively assumed to spend 80% of their time in their burrows.

TRVs were therefore estimated using Equation 7-6:

$$Benchmark = NOAEL \text{ or } LOAEL \times (hr/d \text{ } CF) \times (d/wk \text{ } CF) \times (\% \text{ } t \text{ } CF) \tag{7-6}$$

where
NOAEL	=	No observed adverse effect level
LOAEL	=	Lowest observed adverse effect level
hr/d CF	=	Conversion factor for converting the hours per day of exposure in the toxicity study to receptor exposure
d/wk CF	=	Conversion factor for converting the days per week of exposure in the toxicity study to receptor exposure
% t CF	=	Conversion factor for estimating the time the receptor is expected to spend in the burrow, conservatively estimated at 80%.

The low and high TRVs selected for assessment of risk from inhalation exposure are presented in Table 7-23. Of the 28 VOCs historically detected in soil gas at IR Site 2, TRVs were selected for 22 compounds. NOAELs were not available for naphthalene, toluene, and trichloroethene, but LOAELs were available. The NOAELs for these compounds were estimated by dividing the LOAEL by an uncertainty factor of 10. The NOAEL and LOAEL for 1,4-dichlorobenzene were used to evaluate risk from exposure to 1,3-dichlorobenzene. The low benchmark for total xylenes was used as a surrogate for evaluating *m*-, *o*-, and *m,p*-xylenes. Chronic or subchronic inhalation toxicity data from studies conducted on small mammals were not readily available for 1,1-DCA, *n*-propylbenzene, PCE, trichlorofluoromethane, 1,2,4-trichloromethane, and 1,3,5-trichloromethane. However, these compounds were not detected at a high frequency in site soil gas samples (see Section 5.3.1.4 for a discussion of the nature of contamination detected in soil gas at the site).

7.3.2.1.2   Wetland

Wetland plants and terrestrial invertebrates benchmarks are based on the same sources as those discussed in Section 7.3.2.1.1 for the upland plants and invertebrates. There are not widely accepted benchmarks specific to wetland plant and invertebrate receptors; therefore, the aforementioned benchmarks were used in the absence of more habitat-specific values.

Wetland Ponds

The wetland ponds portion of IR Site 2 is considered an aquatic area with sediment-associated and aquatic-associated receptors that may be exposed to site-related COPECs via direct contact to COPECs in sediment and surface water. These receptors include benthic invertebrates, aquatic invertebrates, and fish. Recent salinity data collected via water quality meters deployed in both the North and South Ponds over a period of approximately 8 months indicate that the ponds can be characterized as saline or hypersaline environments (see water quality data figures in Appendix I). Therefore, marine benchmarks were selected as being the most relevant for conditions in the ponds. Screening-level benchmarks, which correspond to the low benchmarks presented in Table 7-22, are considered appropriate for use in evaluating screening-level effects to these receptors and are described in the following subsections.

Benthic Invertebrates

Screening-level benchmarks for benthic invertebrates in marine sediments were selected in the following order of priority from the following references:

1. Long, E.R., D.D. MacDonald, S.L. Smith, and F.D. Calder. 1995. "Incidence of Adverse Biological Effects Within Ranges of Chemical Concentrations in Marine and Estuarine Sediments." Env. Management, 19:81-97.

2. Long, E.R., and L.G. Morgan. 1991. The Potential for Biological Effects of Sediment-Sorbed Contaminants Tested in the National Status and Trends Program. NOAA Technical Memorandum NOS OMA 52, National Oceanic and Atmospheric Administration.

3. MacDonald, DD, BL Charlish, ML Haines, and K Brydges. 1994. Approach to the Assessment of Sediment Quality in Florida Coastal Waters: Volume 3-Supporting Documentation: Biological Effects Database for Sediment, Florida Department of Environmental Protection, Tallahassee, Fla. *IN* Jones, Suter and Hull. Toxicological Benchmarks for Screening Contaminants of Potential Concern for Effects on Sediment-Associated Biota: 1997 Revision, Prepared for the U.S. Department of Energy.

4. U.S. EPA. 1989b. Evaluation of the Apparent Effects Threshold (AET) Approach for Assessing Sediment Quality, Report of the Sediment Criteria Subcommittee. Science Advisory Board. SAB-EETFC-89-027 *IN* NOAA, National Sediment Quality Survey, Appendix D, Screening Values for Chemicals Evaluated.

Long and Morgan, 1991; and Long et al., 1995

The NOAA effect range-low (ER-L) values for estuarine and marine sediments were selected as screening-level benchmarks to evaluate potential risk to sediment-associated biota. NOAA collected sediment data via a variety of approaches and then ranked chemical concentrations associated with biological effects. ER-Ls represent the low end of the range (lower 10th percentile) of concentrations in marine sediments in which effects were observed or predicted, and are used by NOAA as the concentration below which effects would rarely be observed (Long et al., 1995; Long and Morgan, 1991).

MacDonald et al., 1994

The Florida Department of Environmental Protection (MacDonald et al., 1994) developed marine threshold effect levels (TELs) and probable effect levels (PELs) using the same updated and revised data set used by Long et al. (1995). However, the TELs and PELs also incorporate chemical concentrations observed or predicted to be associated with no adverse biological effects data (no effects data). Specifically, the TEL is the geometric mean of the 15th percentile in the effects data set and the 50th percentile in the no effects data set (Jones et al., 1997). As a result, the TEL represents the upper limit of the range of sediment contaminant concentrations dominated by no effects data. The TEL was used as the screening-level benchmark in cases where an ER-L was not available.

U.S. EPA Apparent Effect Thresholds (AETs)

AETs were used as benchmarks in situations where neither a NOAA ER-L nor a FDEP TEL were available. The AET approach uses data from matched sediment chemistry and biological effects measures and reports sediment concentrations above which statistically significant biological effects

always occur. This concentration is identified as a high no effect concentration (NEC). AETs are used for preliminary comparisons to give an indication of the magnitude of contamination, but are only used in cases where other benchmarks are not available.

Fish and Water Column Invertebrates

Screening-level benchmarks for marine fish and water column invertebrates were gathered from the following references:

1. U.S. EPA. 2004. National Recommended Water Quality Criteria; 2004. Washington, DC; Office of Water and Office of Science and Technology. http://www.epa.gov/waterscience/criteria/nrwqc-2004.pdf

2. U.S. EPA. 2000b. Water Quality Standards: Establishment of Numeric Criteria for Priority Toxic Pollutants for the State of California. 40 Code of Federal Regulations (CFR), Part 131.

3. Buchman, M.F. 1999. NOAA Screening Quick Reference Tables (SQuiRT), NOAA HazMAT Report 99-1. Seattle, WA: Coastal Protection and Restoration Division, National Oceanic and Atmospheric Administration.

U.S. EPA, 2000 and 2004

Low benchmarks for aquatic life are the lowest (and thereby most conservative) of the chronic state (U.S. EPA, 2000b) and federal ambient salt water quality criteria (AWQC; U.S. EPA, 2004). Water quality standards protective of aquatic life were established for 23 priority toxic pollutants in the State of California in the California Toxics Rule (U.S. EPA, 2002). The federal AWQC developed by the U.S. EPA were to be protective of aquatic life, including invertebrates, aquatic plants and fish (U.S. EPA, 2002). In the case of *gamma*-benzene hexachloride, only an acute AWQC value was available; therefore, chronic toxicity was estimated by dividing the acute value by 10.

Buchman, 1999

Marine acute and chronic values presented in the NOAA Screening Quick Reference Tables (SQuiRTs) were used for constituents when a benchmark was not available from any of the other benchmarks sources. Chronic values were selected as screening-level benchmarks where available. Consistent with U.S. EPA guidance on the use of acute to chronic conversion factors, acute values were converted to chronic values by applying a conversion factor of 10 when a chronic value was not available. This conversion was necessary for the following chemicals: chromium, thallium, 2,4-dinitrophenol, 2-methylnaphthalene, 4-nitrophenol, anthracene, benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluroanthene, benzo(g,h,i)perylene, benzo(k)fluoranthene, chrysene, dibenzo(a,h)anthracene, fluorene, hexachlorobutadiene, hexachlorocyclopentadiene, hexachloroethane, indeno(1,2,3-cd)pyrene, isophorone, naphthalene, nitrobenzene, phenol, pyrene, 1,1,1-trichloroethane, 1,1,2,2-tetrachloroethane, 1,2-dichloroethane, carbon tetrachloride, ethylbenzene, trichloroethane, and 4,4-DDD.

7.3.2.2   Food Web Toxicity Reference Values (TRVs)

A key component of a wildlife risk assessment is the identification of appropriate TRVs. A TRV is an exposure level that is considered not to pose unacceptable risk of deleterious effects to a receptor class. TRVs may be developed for different routes of exposure such as ingestion, inhalation, or dermal contact. They may be obtained from appropriate regulatory criteria or be developed as an exposure dose (expressed as mg/kg-BW/day for oral intake), a concentration in food, water, or air (expressed as mg/kg,

mg/L, and milligram per cubic meter [mg/m^3], respectively), or a body burden. This assessment focuses on food-chain exposure and incidental ingestion of soil and sediment, so TRVs were selected from studies that were based on ingestion of COPECs. Screening-level TRVs for wildlife, used in the following order of priority, are presented in Table 7-24:

1. BTAG TRVs: TRVs developed by the U.S. EPA's Region 9 Biological Technical Assistance Group (BTAG) were selected when available for use in the ERA (DTSC, 2001).

2. U.S. EPA Eco-SSL TRVs: When BTAG TRVs were not available, NOAEL values developed by U.S. EPA to Eco-SSLs (U.S. EPA, 2005) were used.

3. ORNL TRVs: If TRVs were not available for a COPEC from either BTAG or U.S. EPA Eco-SSL sources, TRVs developed for ORNL (Sample et al., 1996) were used.

BTAG TRVs

These values are based on a review and interpretation of the body of available literature for each COPEC. Thus, values selected by the BTAG are considered protective of a wide range of avian and mammalian species. Low and high TRVs were developed by the BTAG, with low TRVs reflective of a NOAEL and high TRVs reflective of a LOAEL.

U.S. EPA Eco-SSL TRVs

An extensive list of chemicals was analyzed in site media, and TRVs have not been established by the U.S. EPA Region 9 BTAG for many of them. The next preferred sources for screening-level TRVs were NOAEL values developed for U.S. EPA Eco-SSLs (U.S. EPA, 2005). Available NOAELs from U.S. EPA Eco-SSLs were used as low TRVs for a number of chemicals lacking BTAG TRVs. Most U.S. EPA Eco-SSL LOAEL values were determined by taking the geometric mean of the LOAEL values from the growth and reproduction studies listed in the Eco-SSL documents for the specific constituent. Other U.S. EPA Eco-SSL LOAEL values were taken from a single study. The method for LOAEL development is noted in the following subsections.

ORNL TRVs

If TRVs were not available for a COPEC from either BTAG or U.S. EPA Eco-SSL sources, TRVs developed for ORNL (Sample et al., 1996) were used. For chemicals that lacked TRVs from these three sources, NOAEL and LOAEL values derived from published literature that have applied previously in other ERAs conducted for the Navy or at similar sites in California were used. In some cases, TRVs for an appropriate surrogate chemical were applied. The specific TRVs that were selected for use in the ERA and the corresponding references are listed in Table 7-22 including TRVs used to evaluate risk from TEQ-based doses. Note that Table 5-3 presented the PCDD/PCDF TEFs that were used to account for the relative toxicity of individual PCDD/PCDF congeners in the exposure assessment. The following sections provide additional discussion and rationale regarding TRV selection.

Other TRV Sources

Chlordane: An avian NOAEL for chlordane was proposed in comments from USFWS on the Final RI Sampling Work Plan (Battelle et al., 2005). The proposed value of 0.0014 mg/kg-day appears to have been calculated from data in a study cited as Biotox (1969) in a National Resource Council of Canada document from 1975. The Biotox (1969) study is not available in the published literature, and the value proposed in the comments cannot be reproduced. The value proposed by USFWS is

three orders of magnitude lower than the avian NOAEL of 2.1 mg/kg-day calculated by Sample et al. (1996) based on a study of red-winged blackbirds. The proposed value was used in the screening-level evaluation of avian exposure to chlordane. The LOAEL for chlordane from Sample et al. (1996) was selected for use in the BERA. These low and high chlordane TRVs were used for evaluation of the following components of chlordane: *alpha*-chlordane, *gamma*-chlordane, *trans*-nonachlor, and *cis*-nonachlor.

Explosives: When available, low and high TRVs for explosives for avian and mammalian receptors developed by the U.S. Army Center for Health Promotion and Preventive Medicine (USACHPPM) were selected for use in the ERA (USACHPPM, 2001a and 2001b; and Salice and Holdsworth, 2001). Avian TRVs were available for RDX and 2,4,6-trinitrotoluene. Mammalian TRVs were available for HMX, RDX, 1,3,5-trinitrobenzene, 1,3-dinitrobenzene, 2,4,6-trinitrotoluene, 4-amino-2,6-dinitrotoluene, and 2-amino-4,6-dinitrotoluene. The U.S. Army values were developed from the literature using a protocol described in Standard Practice for Wildlife Toxicity Reference Values (USACHPPM, 2000).

7.3.3 Screening-Level Risk Estimates

The Tier 1 assessment was conducted specifically to determine whether COPECs are present in site media (i.e., soil, sediment, or surface water) at concentrations that could potentially pose risk to ecological receptors at the site. This assessment determined which COPECs in each medium require further evaluation in the Tier 2 BERA. To conduct this Tier 1 analysis, COPECs were identified and conservative EPCs (e.g., maximum detected concentrations or modeled daily doses based on maximum detected concentrations) for each COPEC and receptor were compared to highly conservative effect levels (i.e., NOAEL-based toxicity benchmarks) to calculate HQs. The Tier 1 analysis was designed to ensure that no COPECs that could potentially pose risk were eliminated at this stage. Multiple constituents were unable to be evaluated completely in Tier 1 due to a lack of effects data. These compounds were not screened out and were carried forward to the Tier 2 BERA. The following sections present the results of the Tier 1 assessment for terrestrial, wetland, and wetland pond receptors, and the results are summarized in Tables 7-25 through 7-37. Receptor-specific dose and HQ calculations sheets are provided in Appendix K-2.

7.3.3.1 Upland

The screening-level evaluation for the upland habitat includes an assessment of risk to a number of assessment endpoints which address upland plants and invertebrates, birds, and mammals.

The constituents that were not detected in the top foot of the upland soils, and are therefore not considered as COPECs for invertebrates, birds, or mammals in this evaluation, include 26 of 71 measured SVOCs/PAHs and TPHs, 48 of 52 measured VOCs, 3 pesticides (endrin, endrin ketone, and toxaphene) and 9 of 11 explosives constituents.

7.3.3.1.1 Upland Plants

The screening-level evaluation for upland plants assessed COPECs in the 0 to 6 foot depth strata. COPECs that could not be evaluated due to a lack of effects data, and those chemicals that exceed screening-level thresholds were carried forward to the BERA. Table 7-25 presents the EPCs, benchmarks, and resultant hazard quotients for the screening-level evaluation. Those constituents that were detected but could not be evaluated due to the lack of plant benchmarks include 23 SVOCs/PAHs and TPHs, 18 VOCs, all pesticides, tributyltin, iron, PCDD/PCDF TEQ, and explosives constituents.

COPECs with screening-level hazard quotients exceeding one include a number of metals and PAHs. These constituents will be evaluated further in the BERA.

COPECs with screening-level HQs less than one include arsenic, barium, beryllium, selenium, thallium, 1,1-biphenyl, acenaphthylene, dibenz(a,h)anthracene, diethyl phthalate, di-*n*-butyl phthalate, pentachlorophenol, phenol, styrene, toluene, and PCBs. These constituents were not evaluated further in the BERA.

7.3.3.1.2   Upland Invertebrates

Table 7-26 presents the EPCs, benchmarks, and resultant hazard quotients, where available, for constituents detected in upland surface soils. Those constituents that were detected but could not be evaluated due to the lack of an upland invertebrate benchmark include 9 metals, 24 SVOCs/PAHs and TPHs, 4 VOCs, all pesticides, tributyltin, PCDD/PCDF TEQ, and a single explosives constituent.

COPECs with HQs exceeding one include 6 metals, with chromium and mercury exceeding the benchmarks by the greatest magnitude. A number of SVOCs/PAHs also have HQs greater than one. These constituents were evaluated further in the BERA.

COPECs with screening-level HQs less than one include antimony, arsenic, beryllium, cadmium, lead, nickel, selenium, acenaphthene, acenaphthylene, benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(k)fluoranthene, chrysene, dibenz(a,h)anthracene, dimethyl phthalate, fluoranthene, indeno(1,2,3-c,d)pyrene, n-nitrosodiphenylamine, pentachlorophenol, and phenol. These constituents will not be assessed further in the BERA.

7.3.3.1.3   Upland Birds

Three upland birds were evaluated as part of the upland screening-level assessment. Potential risks to upland birds were evaluated through: (1) exposure to COPECs through the food chain; (2) external exposure to radioisotopes; and (3) exposure to VOCs in burrow air (inhalation TRVs were only available for mammals, so a direct evaluation of birds could not be conducted). Potential risk to birds is discussed based on results of the mammal assessment. Table 7-27 presents the screening-level HQ values for the upland avian receptors. Table 7-28 summarizes the results of the RESRAD-BIOTA screening-level evaluation of risk to upland receptors from maximum concentrations of radionuclides detected in soil. Table 7-29 presents the results of the screening-level risk evaluation to burrowing mammals from inhalation of VOCs historically detected in soil gas at the site.

Food Chain Evaluation

The screening-level assessment resulted in HQs greater than one for eight metals (cadmium, chromium, copper, lead, mercury, nickel, vanadium, and zinc), LPAHs, total DDT, total PCBs, and PCDD/PCDF TEQ. The western meadowlark was generally the most sensitive avian receptor. There was a lack of effects data to evaluate the VOCs and explosives constituents, so the detected VOCs (MBIK, acetone, and methylene chloride) and explosives (1,3 dinitrobenzene and 2,4 dinitrotoluene) were carried forward and evaluated in the BERA.

Radionuclide Evaluation

Table 7-28 summarizes the results of the RESRAD-BIOTA screening-level evaluation of risk to upland receptors from maximum concentrations of radionuclides detected in soil. Although Pb-210 was detected in upland soil samples, it is not included in the RESRAD-BIOTA calculator and could not be included in the risk evaluation. The sum of the ratios (maximum soil concentration/BCG) for

the five radionuclides evaluated is 0.108. Because the sum of the ratios is less than one, radionuclides were not carried forward to the BERA for the upland area.

Inhalation of Burrow Air

Inhalation TRVs were not available for avian receptors. It is not known whether birds are more or less sensitive than mammals to exposure by inhalation. If it is assumed that the sensitivities of birds and mammals are similar and that burrowing birds do not spend any more time in burrows than mammals do, the level of risk to birds would be similar to that observed for mammals. For mammals, HQs were less than one for all but one of the 23 chemicals that could be evaluated. Only the HQ for naphthalene exceeded one. The maximum naphthalene concentration significantly exceeded all other reported concentrations in soil gas at the site since March 2003. Inhalation exposure to naphthalene is evaluated further in the BERA.

7.3.3.1.4 Upland Mammals

Three upland mammals were chosen as ingestion-based exposure receptors. Potential risks to upland mammals were evaluated through: (1) exposure to COPECs through the food chain; (2) external exposure to radioisotopes; and (3) exposure to VOCs in burrow air. Table 7-27 presents the screening-level HQs for the upland mammalian receptors, and Appendix K-2 presents the detailed dose and HQ calculations for upland mammals. Table 7-28 summarizes the results of the RESRAD-BIOTA screening-level evaluation of risk to upland receptors from maximum concentrations of radionuclides detected in soil. Table 7-29 presents the results of the screening-level risk evaluation to burrowing mammals from inhalation of VOCs historically detected in soil gas at the site.

Food Chain Evaluation

The screening-level assessment showed HQs greater than one for nine metals (cadmium, chromium, copper, lead, manganese, molybdenum, nickel, selenium, and zinc), dieldrin, and PCDD/PCDF TEQ. There was a lack of effects data necessary to evaluate the VOC and explosive constituents, so the detected VOCs (MBIK, acetone, and methylene chloride) and explosives (1,3 dinitrobenzene and 2,4 dinitrotoluene) were carried forward and evaluated in the BERA.

Screening-level HQs were less than one for mammalian receptors for 9 metals (arsenic, barium, beryllium, chromium, cobalt, manganese, mercury, selenium, and vanadium), total PCBs, and total DDT. These COPECs were not carried forward to the BERA.

Radionuclide Evaluation

Table 7-28 summarizes the results of the RESRAD-BIOTA screening-level evaluation of risk to upland receptors from maximum concentrations of radionuclides detected in soil. Although Pb-210 was detected in upland soil samples, it is not included in the RESRAD-BIOTA calculator and could not be included in the risk evaluation. The sum of the ratios (maximum soil concentration/BCG) for the five radionuclides evaluated is 0.108. Because the sum of the ratios is less than one, radionuclides were not carried forward to the BERA for the upland area.

Inhalation of Burrow Air

Table 7-29 presents the results of the screening-level risk evaluation to burrowing mammals from inhalation of VOCs historically detected in soil gas at the site. Maximum soil gas concentrations were used to calculate HQs. HQs were less than one for all but one (naphthalene) of the 23 chemicals that could be evaluated. The maximum naphthalene concentration significantly exceeded all other

reported concentrations in soil gas at the site since March 2003. Inhalation exposure to naphthalene was evaluated further in the BERA.

7.3.3.2   Wetland

The screening-level evaluation for the wetland habitat includes an assessment of risk to wetland plants and invertebrates, birds, and mammals. The constituents that were not detected in the wetland surface soils (0-1 ft bgs) and are therefore not considered as COPECs for the screening-level evaluation of risk to wetland invertebrates, birds and mammals include molybdenum, 34 of 68 SVOCs/PAHs and TPHs, 45 of 52 VOCs, and 10 pesticides.

7.3.3.2.1   Wetland Plants

Table 7-30 presents the EPCs, benchmarks, and hazard quotients where available for constituents detected in wetland soils (0-TD). Those constituents that were detected but could not be evaluated due to the lack of a benchmark include 15 SVOCs/PAHs and TPHs, 10 VOCs, all pesticides, PCDD/PCDF TEQ, iron, magnesium, and tributyltin.

Those constituents with HQs exceeding one include 13 different metals with chromium, vanadium, and zinc exceeding benchmarks by the greatest magnitude, and all SVOCs/PAHs. These constituents were evaluated further in the BERA.

COPECs with screening-level HQs less than one include antimony, barium, beryllium, cadmium, hexavalent chromium, thallium, 1,1-biphenyl, acenaphthene, acenaphthylene, anthracene, benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(g,h,i)perylene, benzo(k)fluoranthene, chrysene, dibenz(a,h)anthracene, diethyl phthalate, di-*n*-butyl phthalate, fluoranthene, fluorine, indeno(1,2,3-cd)pyrene, naphthalene, pentachlorophenol, phenanthrene, phenol, pyrene, and PCBs. These COPECs were not evaluated further in the BERA.

7.3.3.2.2   Wetland Invertebrates

Table 7-31 presents the EPCs, benchmarks, and hazard quotients where available for constituents detected in wetland surface soils (0-1 ft bgs). Those constituents that were detected but could not be evaluated due to the lack of an invertebrate benchmark include 9 metals, 13 of 68 SVOCs/PAHs and TPHs, 6 VOCs, all pesticides, and PCDD/PCDF TEQ.

Those constituents whose HQs exceed one include 4 metals (chromium, copper, mercury, and zinc); therefore, these constituents were evaluated further in the BERA.

COPECs with screening-level HQs less than one include antimony, arsenic, barium, beryllium, cadmium, hexavalent chromium, lead, nickel, selenium, all SVOCs/PAHs, and chlorobenzene. These constituents therefore were not evaluated further in the BERA.

7.3.3.2.3   Wetland Birds

Four wetland birds were chosen as ingestion based exposure receptors in the wetland area. Potential risks to wetland birds were evaluated through: (1) exposure to COPECs through the food chain; (2) external exposure to radioisotopes; and (3) exposure to VOCs in burrow air. Although exposure to burrow air is an identified pathway, soil gas data from the wetland area does not exist, thus that pathway could not be quantitatively evaluated. Table 7-32 presents the screening-level HQs, and Appendix K-2 presents the detailed dose and HQ calculations for wetland birds. Table 7-33 summarizes the results of the RESRAD-